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On all editorial matters, please contact Michael Huft, 232 Akela Dr., Valparaiso, IN 46385. Phone: (847) 682-5240; email: mhuft@att.net. All articles dealing with botany in the Great Lakes region may be sent to the Editor at the above address. In preparing manuscripts, authors are requested to follow the "Instructions for Authors" on the inside back cover.

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IN THIS ISSUE

The preservation of natural ecosystems has necessarily become a topic of increasing concern in recent decades. The two principal articles in this issue deal with this common concern in two very different ways. The first article takes seriously the threat posed by aggressive invasive plants. Most frequently, an invasive is a non-native species unrelated to plants in the native habitat. In this case, however, the threat is particularly pernicious, because the invasive element is an introduced genotype of a species already present as a native and posing no threat to the habitat in question, that is, common reed, *Phragmites australis*. The authors address the preliminary process of determining the presence of the different genotypes in habitats along the West Coast of Michigan and their distribution. They also discuss the factors that might contribute to an increase in the presence of the invasive genotype and propose certain control measures.

In an era of massive habitat destruction, the restoration of natural ecosystems has become an important aspect of overall preservation. The second article, based on a Master's thesis, presents a thorough study of the effect of introducing species with various ecological roles in the restoration of a sand prairie, an ecosystem that was once common in the Upper Great Lakes region, but has now been reduced to vanishing remnants. The goal of the study is to determine whether the introduction of certain mixes of species in varying quantities has an effect on the diversity and floristic quality of the restored prairie while at the same time reducing the impact of non-native invasives.

In this issue, we are making a change in the format for "Noteworthy Collections" articles, following a suggestion by a contributor. "Noteworthy Collections" will be a section of the journal, and each article in the section will have its own title. That way, the articles can more readily be cited and listed in bibliographies in a way that makes their content apparent. Although it happens that each of the Noteworthy Collections articles in this issue treats a single species, that does not mean that articles submitted for this section with more than one item need to be divided into separate articles, each with its own title. Multiple-item articles may carry relevant titles, such as "Five New Reports for Michigan," or whatever might be appropriate. The two articles in this issue confirm the presence in western Michigan of American Lotus, not seen there since 1942, and report the first occurrence in Michigan of a non-native species that has been spreading in North America for some time.

This issue closes with a review of two excellent field guides to a substantial portion of the graminoids of Wisconsin that should be useful throughout the western Great Lakes region.

——Michael Huft

A SURVEY OF NATIVE AND INVASIVE GENOTYPES OF *PHRAGMITES AUSTRALIS* ALONG MICHIGAN'S WEST COAST

Grady H. Zuiderveen[1]
Department of Ecosystem Science and Management
Pennsylvania State University
State College, Pennsylvania 16802

Timothy M. Evans, Thomas J. Schmidt, and Mark R. Luttenton
Biology Department
Grand Valley State University
Allendale, Michigan 49401

ABSTRACT

In North America, *Phragmites australis* (common reed) consists of a native North American group with several genetic forms and a highly invasive group with a single genetic form. Little is known about the environmental factors that affect the relative distribution of native and invasive populations, and it is often difficult to differentiate between the two groups based on physical characteristics alone. The western Lower Peninsula of Michigan provides an excellent opportunity to evaluate the relationship between environmental gradients and the spread of *Phragmites*. The Lake Michigan coast presents a strong temperature gradient from the north to the south, whereas a moisture gradient is found from west to east from the Lake Michigan shoreline inland. The goal of this study is to assess the risk of invasion by the invasive genotype by: 1) determining the genotypes of individuals in *Phragmites australis* populations in western Michigan; and 2) evaluating the geographic distribution of the representative genotypes throughout the region. Examination of 58 samples of *Phragmites australis* using chloroplast gene markers from throughout western Lower Peninsula of Michigan yielded only the invasive genotype in the southern regions and a mixture of native, invasive, and Gulf Coast genotypes in the northern regions. Only one of the populations examined contained individuals of both native and invasive genotypes. No trends were observed along the east-west gradient.

KEYWORDS: Invasive genotype, *Phragmites australis*, Relative distribution.

INTRODUCTION

Wetlands are a critical habitat component of the Great Lakes region and serve several important functions, including wildlife habitat, sediment retention, and water purification (Keddy 2000). Anthropogenic disturbance (draining or filling) has led to substantial declines in the number and size of Michigan's wetlands. Exotic invasive species, particularly invasive populations of *Phragmites australis* (common reed), have emerged as a significant threat to both coastal and non-coastal wetlands throughout the region. *Phragmites australis* is a perennial wetland grass that can be found in both brackish and freshwater habitats (Saltonstall,

[1]Author for correspondence (gjz5033@psu.edu)

2003a). It is found worldwide in marsh communities; along rivers, lakes, and ponds (Saltonstall 2002); and in roadside ditches. Currently, it is considered the most invasive plant species in wetland regions of the east coast of the United States (Rudrappa et al. 2007).

Human factors known to aid in the expansion of the range of *Phragmites* include ditch-digging for mosquito control and improved farm drainage (thereby lowering the water table), nutrient enrichment caused by runoff from lawns and farms (Amsberry et al. 2000), and possibly the transport of soil containing vegetation fragments or seeds (e.g., Keller 2000). Historically, native *Phragmites* comprised only a small component of the vegetation of coastal marshes (Peter and Burdick, 2010), where it was restricted to the higher border regions, but the invasive *Phragmites* has more recently been noticeably invading the lower, wetter areas (Amsberry et al. 2000).

North American *Phragmites* currently consists of a native North American group with several genetic forms (based on plastid genotypes) and an invasive group with one genetic form (*sensu* Saltonstall 2002) that is extremely aggressive and that spreads quickly throughout wetlands. For example, Whyte et al. (2008) reported that in 1993, *Phragmites australis* occupied less than 1% of the lower wetland at the Old Woman Creek National Estuarine Reserve in Ohio and roughly 3% in 2003. By 2005, *P. australis* dominated 22% of the lower wetland and was also a substantial component of the emergent plant community. *Phragmites australis* has gone from being a minor component of marshes to being a nuisance species as the native genotype has been replaced by the invasive genotype, resulting in significant ecological impacts (Peter and Burdick 2010). In some wetlands, the spread of invasive *Phragmites* has eliminated much of the native vegetation and converted diverse wetland plant communities to monospecific stands of common reed.

The ability of *Phragmites* to colonize open areas rapidly is attributed to effective methods of natural reproduction by the spread of hardy seeds and rhizomes (Saltonstall 2003b), through human practices, and by the dispersal of rhizome fragments. Another reason that *Phragmites* has been so successful is its clonal architecture of interconnected culms spanning over 10 meters (Burdick and Konisky 2003). Furthermore, some findings indicate that *Phragmites* exudes gallic acid, which generates high levels of reactive oxygen species in other plants, resulting in increased mortality (Rudrappa et al. 2007). It should be noted, however, that in a more recent study, Weidenhamer et al. (2013), refuted that claim, since they were unable to find significant amounts of gallic acid in or around invasive *Phragmites*.

Phragmites now occupies significant portions of Great Lakes coastal wetlands, including those along the eastern shore of Lake Michigan, and of isolated non-coastal wetlands. Although previous studies have been conducted to characterize the distribution of *Phragmites* genotypes across North America (Saltonstall 2003a), it appears that little has been reported about *Phragmites* populations in the upper Great Lakes region. It is therefore unclear which genotypes are present throughout the region. If the invasive genotype has become established, then the native genotypes of *P. australis*, as well as the coastal and non-coastal wetlands subject to invasion by the invasive genotype, are at risk, thereby affecting

native plant and animal species (Whyte et al. 2008). The goal of this study was to assess this risk by: 1) determining the genotype of individuals in *Phragmites australis* populations in western Michigan; and 2) evaluating the geographic distribution of the representative genotypes throughout the study area.

METHODS

A total of 58 samples from 40 populations of *Phragmites* along Michigan's west coast (Figure 1) were collected and georeferenced using a handheld GPS unit. Maps were created in ArcGIwwS 10 (ESRI 2011) using the locations of the georeferenced samples and projected in NAD 1983 State Plane, Michigan south international feet. Multiple samples were collected at 14 of the 40 collection sites where phenotypic (e.g., plant height, growth density) or environmental (e.g., soil moisture, species richness) differences were evident, or where the genet was spread over a large area. When *Phragmites* were restricted to small areas that appeared homogenous, only a single sample was collected as a single genet appeared likely. This sampling approach was designed to maximize efficiency, since the emphasis of this study is to determine a broad-scale pattern as opposed to comprehensively mapping genotype distributions. Samples were collected from coastal and drowned-river mouth wetlands extending from Berrien County in Michigan's southwest corner to Antrim County in the northwest portion of the Lower Peninsula. Additional collections were made from sites that represented an east-west gradient in roadside ditches and wetlands away from the Lake Michigan coast for as far as 65 kilometers (40 miles). Fresh leaf tissue was collected and either frozen at -80°C or desiccated in silica gel. A voucher specimen for each sample was deposited in the Grand Valley State University herbarium (GVSC).

DNA was extracted from leaf tissue using the DNeasy Plant Mini Kit made by Qiagen. Chloroplast genes *trnLb* and *rbcL* were amplified using primers described in Saltonstall (2002). As described in Saltonstall (2002), *trnLb* fragments were digested with *RsaI*, and *rbcL* fragments were digested with *HhaI*, in each case to distinguish invasive from either native or Gulf Coast genotypes.

RESULTS

Of the 58 plant samples that were examined, 47 exhibited the invasive genotype (Figure 1, Table 1). The invasive genotype was distributed throughout the entire range of the study, including at our most northern site. It was the only genotype found in the southern third of the Lower Peninsula, and it was the most common genotype in the study.

Of the remaining 11 plant samples, six, from four populations, exhibited a native genotype. These populations were restricted to the west-central and northern portions of the Lower Peninsula. Five plant samples from four populations exhibited the Gulf Coast genotype (Figure 1, Table 1). Populations that exhibited the Gulf Coast genotype were located in the west-central region of the Lower Peninsula. Of the 40 sites sampled, only two mixed populations were found. Both native and invasive genotypes were found at one site in northwestern Ottawa County, and another site in the same region of northwest Ottawa County contained both the invasive and the Gulf Coast genotypes. At the remaining 38 sites, no other mixed populations were found. This may have been due either to true homogenous strands or to cryptic mixing in which no differences in phenotype or environment were visually identifiable, and thus only one sample was collected.

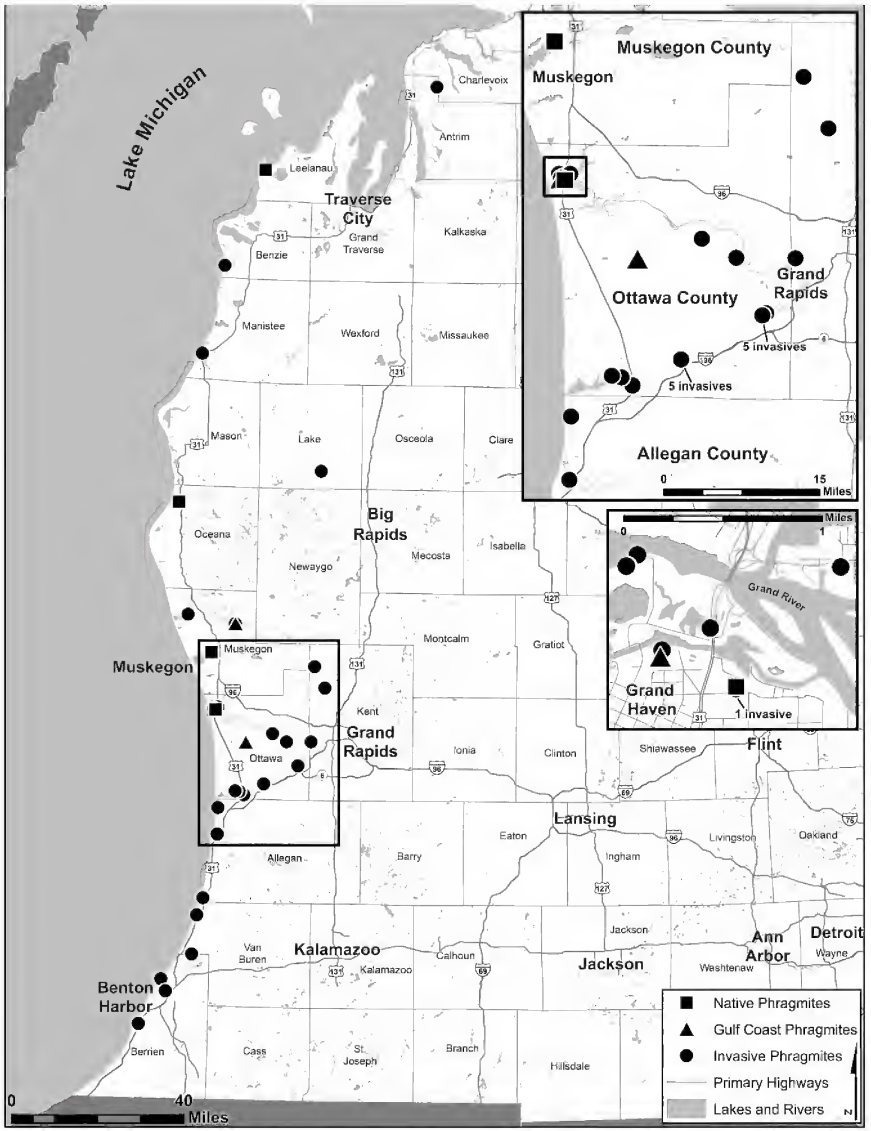

FIGURE 1. Location of sampling sites of *Phragmites australis* in western Michigan. Each point indicates a single individual that was sampled. The collection number, latitude, and genotype of each sample are given in Table 1. In addition, the genotype at each site is indicated on the map by the symbols ■ (native genotype), ▲ (Gulf Coast genotype), and ● (invasive genotype).

TABLE 1. .Collection numbers of samples of *Phragmites australis,* listed in geographic order from south to north, with the latitude (second column) and the genotype (third column) of each collection.

Collection No.	Latitude	Genotype
32	N 42°00.355′	invasive
33	N 42°00.372′	invasive
34	N 42°00.409′	invasive
35	N 42°07.139′	invasive
30	N 42°09.505′	invasive
31	N 42°09.531′	invasive
36	N 42°14.715′	Gulf Coast
27	N 42°22.603′	invasive
26	N 42°26.046′	invasive
25	N 42°39.017′	invasive
24	N 42°44.311′	invasive
21	N 42°47.009′	invasive
22	N 42°47.659′	invasive
23	N 42°47.777′	invasive
9	N 42°49.191′	invasive
10	N 42°49.215′	invasive
11	N 42°49.216′	invasive
8	N 42°49.239′	invasive
12	N 42°49.254′	invasive
58	N 42°52.420′	invasive
59	N 42°52.420′	invasive
6	N 42°53.056′	invasive
7	N 42°53.057′	invasive
3	N 42°53.254′	invasive
2	N 42°53.255′	invasive
1	N 42°53.257′	invasive
49	N 42°57.560′	Gulf Coast
50	N 42°57.590′	Gulf Coast
5	N 42°57.844′	invasive
4	N 42°57.869′	invasive
28	N 42°57.894′	invasive
29	N 42°59.410′	invasive
13	N 43°04.029′	invasive
14	N 43°04.038′	native
52	N 43°04.160′	Gulf Coast
51	N 43°04.190′	invasive
19	N 43°04.291′	invasive
16	N 43°04.553′	Invasive
17	N 43°04.553′	invasive
18	N 43°04.553′	invasive
20	N 43°04.571′	invasive
15	N 43°04.604′	invasive
48	N 43°08.775′	invasive
56	N 43°13.060′	invasive
57	N 43°13.060′	invasive
39	N 43°15.537′	native
40	N 43°15.623′	native
53	N 43°21.450′	Gulf Coast
54	N 43°21.450′	invasive
55	N 43°23.140′	invasive
41	N 43°23.217′	invasive
42	N 43°45.770′	native
43	N 43°45.775′	invasive
44	N 44°15.736′	invasive
60	N 44°33.360′	invasive
46	N 44°52.969′	native
45	N 44°52.984′	native
47	N 45°10.174′	invasive

DISCUSSION

The specific arrival time of the invasive genotype of *Phragmites* to the Midwest is still unknown. Since the morphological characteristics of the invasive genotype are very similar to those of the native genotype, the presence of the invasive genotype likely went unnoticed for a number of years. The invasive *Phragmites* genotype dominates both coastal and noncoastal sample locations (including several wetlands) south of Ottawa County, which suggests that the invasion may have been from south to north in the Lower Peninsula.

Given its broad tolerance and aggressive nature, the invasive genotype of *Phragmites* could eventually dominate most or all of the wetlands in Michigan, regardless of latitude. Wetlands crossed by roads may be most vulnerable to invasion by invasive *Phragmites*. Indeed, drainage ditches along county roads and highways have been found to support robust populations in Michigan and may play a large role in their spread by serving as an initial habitat for invasive *Phragmites* throughout the region (Jodoin et al. 2008; Lelong et al. 2007). Because most of our sample sites were adjacent to roads, it is unclear whether the same pattern applies to wetlands in more isolated areas. However, isolated wetlands may be equally vulnerable to alternate routes of invasion. The location of the Benzie County sample site along an isolated lakeshore, for example, suggests that more isolated areas are not immune from the invasive genotype. In these cases, invasion may be through natural dispersal or human agency. For example, human factors such as shoreline development may play a large role in the spread of invasive *Phragmites*. Silliman and Bertness (2004) found that the removal of woody vegetation along the edges of marshes explained 91% of the variation in *Phragmites* cover among the marshes of Narragansett Bay in Rhode Island. In western Michigan, riparian residents often mow wet areas in an attempt to maintain their view and to control *Phragmites*. However, since *Phragmites* appears to recover more quickly than native wetland species after a mowing (Luttenton, personal observation), this process appears to be more detrimental to other vegetation and could promote the establishment of monospecific stands by *Phragmites*.

Another plausible factor that may have contributed to the increase in invasive *Phragmites* in the Great Lakes region could be the decrease in water levels over the past three decades to levels that are comparable to historic mean lake levels (Great Lakes Environmental Research Laboratory 2016). The previous high water levels may have killed off the woody species in coastal communities, thereby leaving space for species that thrive in standing water. Although yearly water level fluctuations are common in the Great Lakes, the recent decrease in water levels to historic lows would leave large areas of exposed sediments that are ideal for the opportunistic and aggressive *Phragmites* (Whyte et al., 2008). Furthermore, it has been proposed that *Phragmites* may often become established initially in the high marsh and then expand into the lower, less ideal, areas by clonal integration (Amsberry et al. 2000). If this is the case, even relatively small amounts of disturbance, whether caused by humans, by decreased water levels, or by any of a number of other mechanisms, could result in the expansion of already-present *Phragmites*. Because it has become so common in many parts

of the United States, invasive *Phragmites* is considered to be an accurate gauge of wetland disturbance (Saltonstall 2002).

A surprising finding in this study was the presence of the Gulf Coast genotype at several sites. The results of this study might lead one to concluded that the Gulf Coast genotype is as common in western Michigan as the native genotype. However, the incidence of the native genotype is likely to have been underestimated due its greater prevalence in northern Michigan, which is less extensively represented in this study. Previous studies had indicated that the Gulf Coast genotype was confined to the Gulf Coast region (Saltonstall et al. 2004), but it now appears that it has spread northward. The presence of the Gulf Coast genotype may be the result of fill transport from construction. Keller (2000) found two distinct populations of *Phragmites* in New England that were hypothesized to be the result of transport of construction fill during the 1950s. Likewise, it has been hypothesized that the early spread of *Phragmites* was facilitated by boating activities (Chambers et al. 2012), for which the Great Lakes region is renowned.

The presence of two populations in Ottawa County that contain both native and invasive genotypes in single locations raises the question of whether hybridization between native and invasive plants has taken place. Saltonstall (2011) reported that the native lineages may maintain a pure genetic composition even in mixed stands. However, Meyerson et al. (2010) found that in a lab setting, hybridization between native and invasive genotypes of *Phragmites* resulting in viable offspring can take place, which could present further complications if it occurred in the wild. Meyerson et al. (2010) also found that there is considerable overlap in flowering phenology between the native and invasive genotypes, which could facilitate natural hybridization. Earlier, Saltonstall et al. (2004) documented a single instance of hybridization in the field. Hybridization has been shown to result in higher seed viability in the invading hybrids, and the resulting progeny can be the most vigorous in species in which hybridization occurs (Ayres et al. 2008). Further, increased genetic diversity can nullify deleterious alleles by forming new favorable genetic combinations (Saltonstall 2003a), making invasive species even more threatening to natural habitats.

Municipalities and local NGOs have recently instituted programs to control the spread of *Phragmites*, for example, the Ottawa County Invasive *Phragmites* Control Group (Great Lakes *Phragmites* Collaborative 2010) and The Watershed Center Grand Traverse Bay (2016). In Ottawa County, where we have found mixed stands of genotypes, several extensive *Phragmites* populations have recently been eradicated by means of aggressive control measures (Luttenton, personal observation).

It is possible to differentiate the native genotype from the Gulf Coast and invasive genotype based on botanically technical differences in morphological features including ligule, glume, and lemma length (Saltonstall et al. 2004; Judziewicz et al. 2014). However, using these morphological differences would limit Municipalities and NGOs time of treatment to when reproductive parts are present and would require some botanical expertise. Therefore, given the similarity between the native and invasive forms, we suspect that eradication efforts

may not discriminate between them. Consequently, native populations, which pose little threat to native wetland communities, may nevertheless be lost.

CONCLUSION

Invasive *Phragmites* has successfully colonized Michigan wetlands and presents a significant threat to both coastal and non-coastal wetlands. *Phragmites* is known to be a habitat-modifying organism in that its vigorous growth in lower marshes can result in reduced sediment erosion and increased water uptake and transpiration, resulting in a lowering of the water table (Amsberry et al. 2000). Silliman and Bertness (2004) found that *Phragmites* dominance in developed wetlands leads to substantial decrease in plant species richness. This endangers both native plants and the wildlife they support, including migratory birds dependent upon native plants for nesting.

Eradication of the invasive genotype has been difficult, and sites in Grand Haven, Michigan, were found in this study in which, although chemical applications had recently been applied, young shoots of invasive *Phragmites* could nevertheless be observed. Chemical application is a commonly used treatment for eradicating *Phragmites*. However, a single treatment is ineffective in permanently removing *Phragmites* (Warren et al. 2001); therefore, chemical treatment becomes expensive with repeated application, the only known advantage over alternative treatment methods being that it has no major impact on fish populations (Fell et al. 2006). One alternative to a chemical method is biological control as described by Blossey (2003). He claims that the vigorous nature of the invasive *Phragmites* could be due to the lack of herbivory, allowing the plant to reallocate its resources from fighting herbivory toward more rapid and dense growth. Therefore, if a specialized natural herbivore of *Phragmites* were introduced into North America, the competitive edge of having no enemies that the invasive species now enjoys might be removed, ultimately allowing wetlands to be restored to their original state. Such a method would have an inherent risk that the newly released herbivore would consume plants other than the desired target—including the native genotype. However, since no other species of *Phragmites* are found in North America, the likelihood of such nontarget effects is low (Pemberton 2000).

Alternatively, methods of preventing the further spread of invasive *Phragmites* into the more northern regions of Michigan should be seriously considered. One method that has been successful has been to maintain high levels of native plants in regions at risk for the spread of *Phragmites*. Specialist native competitors provide an ecological barrier to the invading *Phragmites* by competing for essential resources and reducing available niches (Peter and Burdick 2010). *Phragmites* tends to colonize disturbed upland boarders of wetlands first, then to expand into the central regions (Warren et al. 2001). Reducing human disturbance and maintaining species richness in Michigan wetlands is an option worth seriously considering, since decisions regarding management of invasive *Phragmites* have long-term repercussions (Ludwig et al. 2003). Future studies

would be needed to further determine the northern extent of the invasive genotype in order to determine where preventive measures might be the most effective as, at least for now, the invasive genotype of *Phragmites*, once established, is here to stay.

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THE INITIAL EFFECTS OF COMMUNITY VARIABLES ON SAND PRAIRIE RESTORATION: SPECIES ESTABLISHMENT AND COMMUNITY RESPONSES

Robert C. Roos
6140 Cottonwood Drive, Suite A
Fitchburg, Wisconsin 53719
robb.roos@cardno.com

Todd A. Aschenbach
Grand Valley State University
1 Campus Drive
Allendale, Michigan 49401
aschenbt@gvsu.edu

ABSTRACT

The tallgrass prairie was one of the most wide-ranging and diverse ecosystems in North America. This diverse ecosystem comprises a mosaic of prairie types that, in northern Lower Michigan, was historically dominated by dry sand prairie. As a result of fire suppression, silvicultural and agricultural activities, and degradation by invasive species, only approximately 4% of the original extent of sand prairie remains intact in the state. Despite the important ecological role and increasing scarcity of sand prairie, restoration and management of this ecosystem has been severely understudied. In order to gain a better understanding of this ecosystem and of potential restoration techniques that might influence its community variables, a sand prairie restoration experiment was established in the pine–oak barrens of northern Lower Michigan to analyze how different seeding treatments affect certain community variables, including vegetative cover, species richness, diversity, and floristic quality. The seeding treatments were varied with respect to seeding concentrations (1,000 seeds/m^2 and 10,000 seeds/m^2); and the inclusion of grasses and/or forbs with diverse ecological characteristics, such as early flowering, late flowering, and nitrogen fixers (i.e., legumes). Measurements of the community variables were taken during each of the first three growing seasons following seeding. In general, treatments that included a high concentration of grass and/or an early season forb component had the greatest overall positive impact on plant community development. These treatments resulted in significantly greater diversity and higher floristic quality, as well as a lower percentage of non-native or invasive cover than other treatments. The benefit of high concentrations of grasses and early season forbs may play a critical role in initial species establishment of a sand prairie restoration due to the facilitative and competitive advantages they may provide in these harsh environments. However, it remains to be seen if these initially successful communities will have continued success over longer periods of time.

KEYWORDS: Sand prairie, Community, Establishment, Restoration.

INTRODUCTION

The tallgrass prairie was one of the most wide-ranging and diverse ecosystems in North America. Today, only approximately 0.1% of its original extent remains, making tallgrass prairie one of North America's most endangered ecosystems (Samson and Knopf 1994). The extensive loss and rapid decline of this

ecosystem can be attributed primarily to European settlers. The dark, rich, tree-free, and easily manipulated Mollic soils of the tallgrass prairie made it a prime target for agricultural use (Howe 1994). This ecosystem has also experienced many other assaults as a result of human development. Fire, which was once commonplace in the Midwestern United States, became actively managed and suppressed (Anderson 1990). Habitat fragmentation, the introduction of invasive non-native species, and the increased establishment of invasive native species also promoted habitat degradation (Noss et al. 1995; Cully et al. 2003).

The reduction of tallgrass prairie and its associated biodiversity has resulted in a substantial loss of ecosystem function. Higher levels of plant diversity are associated with greater ecosystem productivity (Tilman and Downing 1994; Hector et al. 1999; Tilman 1999), nutrient use efficiency (Risser 1988; Tilman 1997), resistance to invasion by exotic species (Tilman and Downing 1994; Kennedy et al. 2002; Pokorny et al. 2005), and resistance to environmental change (Ives et al. 2000). According to the insurance hypothesis of Yachi and Loreau (1999), high levels of biodiversity insures ecosystems against declines in their functioning, because the presence of many species guarantees that some will remain functioning even if others fail (Ives et al. 2000). Ecosystem functions that are associated with tallgrass prairie include carbon sequestration, water filtration, erosion control, soil enhancement, and nutrient cycling (Raison 1979; Seastedt and Knapp 1993; Wedin and Tilman 1990). Since diversity is a beneficial and necessary attribute that drives a fully-functioning, healthy ecosystem, it is useful to examine the factors that impact diversity.

The tallgrass prairie ecosystem comprises a mosaic of prairie types that include xeric, mesic, and wet components that sometimes transition into barrens and savannah ecosystems. Tallgrass prairie historically extended eastward through Indiana and into areas of Michigan, Ohio, and Kentucky. This region is referred to as the prairie peninsula. These fingers of grassland followed areas where climate fluctuated enough to support a mosaic of prairie community types, including oak-pine barrens and oak savanna (Transeau 1935; Anderson 1990). In northern Lower Michigan, north of the tension zone (approximately 43ºN latitude), grassland was predominantly dry sand prairie (McCann 1991; Kost 2004). As a component of these open, upland mosaics, sand prairie was a primary component of roughly 5,000 hectares of northern Lower Michigan's natural landscape in the early to mid-1800s (Comer et al. 1995). This dry grassland is considered the driest ecosystem east of the Mississippi River (Schaetzl and Anderson 2005). Plant species of the sand prairies are similar to those of a xeric tallgrass prairie, but due to water, heat, and nutrient stresses, the sand prairie vegetation is typically shorter in stature and separated by patches of bare ground. The combination of wildfires, dry and well-drained soils, and the harsh frosts that are associated with northern Michigan have historically maintained these ecosystems (Kost et al. 2007).

As a result of fire suppression, silvicultural and agricultural activities, and degradation by invasive species, only approximately 4% of the original extent of sand prairie remains intact in the state (Hauser 1953; Albert and Comer 2008). Consequently, sand prairie is considered one of Michigan's most endangered

ecosystems. Today, less than 200 hectares of high quality sand prairie still exist in Michigan (Kost 2004).

The loss and degradation of the sand prairie has had negative consequences for species that are associated with these ecosystems. Over 25 plant and 30 animal species are dependent on these dry grasslands for either all or part of their lives (Kost 2004). This includes the federally endangered *Lycaeides melissa samuelis* (Karner blue butterfly), a species that depends on *Lupinus perennis* L. Grassland birds have shown a decline that is greater than that of any other group of North American species (Knopf 1994). *Dendroica kirtlandii* (Kirtland's warbler) is a species that depends solely on the matrix of sand prairie and pine barrens comprised of *Pinus banksiana* Lamb. in northern Lower Michigan for survival (Kost et al. 2007). Common northern dry sand prairie species, that nevertheless are state-listed due to the degradation of this community type, include *Festuca altaica* Trin. Ex Ledeb., *Cirsium hillii* (Canby) Fernald, and *Agoseris glauca* (Pursh) Raf..

Despite the important ecological role and increasing scarcity of sand prairie, restoration and management of this ecosystem has been severely understudied. Published research on sand prairies within the eastern prairie peninsula, including the portion in Michigan, is sparse. Instead, the majority of sand prairie literature has focused on prairie regions to the west of the Great Lakes (Gleason 1910; Plumb-Mentjes and Center 1990; Cole and Taylor 1995; Bowles et al. 2003). Studies that have addressed the Michigan sand prairies have focused on descriptive analyses (Hauser 1953; Albert 1995; Comer et al. 1995; Kost 2004), or on comparative assessments with other community types, such as dunes and jack-pine barrens (Houseman and Anderson 2002; Emery et al. 2013). To date there have been few, if any, studies that have focused on the restoration of Michigan's true sand prairies.

Restoration projects in general tend to place an emphasis on many aspects of community structure and ecosystem processes. Although certain components of restoration projects have been successful, they have yet to achieve the goal of creating a historic, natural community (Martin et al. 2005). Current methods of restoring prairie communities are based on a weak scientific rationale that is not consistent with the history of how grasslands formed, and in fact may threaten biodiversity (Howe 1994). Although plant community restoration has the potential to help re-establish lost diversity and ecosystem function (Foster et al. 2007), many plant community restoration attempts have not fully re-established the diversity and function found in remnant prairie communities (Sluis 2002, Polley et al. 2005).

Sub-optimal results of previous restoration attempts have led ecologists to examine different theories of plant community succession and species coexistence in order to identify more successful approaches to restoration (Cairns and Heckman 1996). Community assembly theory is one approach that asks how species arriving at a site form an initial community (Belyea and Lancaster 1999). This approach integrates aspects of succession and species coexistence in an effort to examine how species introductions, biotic interactions (e.g., competition), and abiotic conditions (e.g., soil nutrients) influence community development (Lockwood 1997; Belyea and Lancaster 1999; Young et al. 2001).

The influence of environmental factors on plant communities is complex, and multiple factors influence plant diversity and community development (Grace 1999). It is well established that the environment influences plant community succession (Tilman 1988; Howe 1995), the frequency and intensity of disturbance (Collins et al. 1995; Suding 1999; Collins 2000), competitive intraspecific and interspecific interactions and the plant's ability to respond (Grime 1974; Goldberg and Barton 1992; Smith et al. 1999), the availability of nutrients (Raison 1979; Ojima et al. 1994), and the ability of plants to be productive both vegetatively and reproductively (Zimmerman and Kucera 1977; Gough et al. 1994; Tilman et al. 1996). A better understanding of how all of these factors influence community development will allow for more practical approaches to community restoration.

As one approach to understanding that question, we established a restoration experiment on a degraded site previously occupied by sand prairie in northern Lower Michigan in 2009. We introduced various combinations of native plants belonging to four functional groups—legumes, early flowering forbs, late flowering forbs, and warm-season grasses—at two different seeding concentrations in an attempt to see how these initial seeding treatments affect plant community development over time. Here we present the results obtained after two complete growing seasons and compare how the different seeding treatments have affected vegetative cover, species richness, diversity, and floristic quality in the harsh environment of a Michigan sand prairie.

METHODS

Study Area

The study site is located at the historic Chittenden Nursery in the Manistee-Huron National Forest in Manistee County, Michigan. The nursery site is approximately 23 hectares and consists of 13 adjacent 1- to 2-hectare open fields. The location chosen is an appropriate study site, because the area was relatively homogenous inthat it was level, tree-free, and dominated by invasive native and non-native species. The rectangular study area covers an approximate 500 m^2 portion of one of these fields. The site location is depicted in Figure 1.

The study site was historically part of the oak-pine barrens ecosystem which included pockets of sand prairie (Albert and Comer 2008). In 1934, the area was converted into the Chittenden Nursery, a tree nursery for the United States Forest Service (USFS). The tree nursery was shut down in the 1970s and has since been used rather insignificantly for USFS housing, conferences, and training activities (e.g., wildfire training, prescribed burns, all-terrain vehicle instruction).

Climatic factors at the time of the study were consistent with historical averages. Summer temperature highs during the survey years averaged 23.1°C, and average monthly rainfall was 17.5 cm. Site soils are mapped as Plainfield sands with a very deep water table (United States Department of Agriculture 2012). The weather and soil conditions are consistent with oak-pine barren and sand prairie ecosystems, and the vegetation is characteristic of a degraded ecosystem (Albert and Comer 2008).

Experimental Design

Prior to the initiation of our experiment, the entire study area was mowed and foliar herbicide containing glyphosate was applied to it in 2007 and again in 2008 in an attempt to reduce the abundance of invasive species (e.g., *Centaurea stoebe* L.).

In March 2009, a total of 228 1 m^2 treatment plot boundaries were established. All plots were separated by a 0.5 m border along all four sides to avoid edge effects and to prevent trampling while taking measurements. Typical sand prairie species used in local U.S. Forest Service restorations were

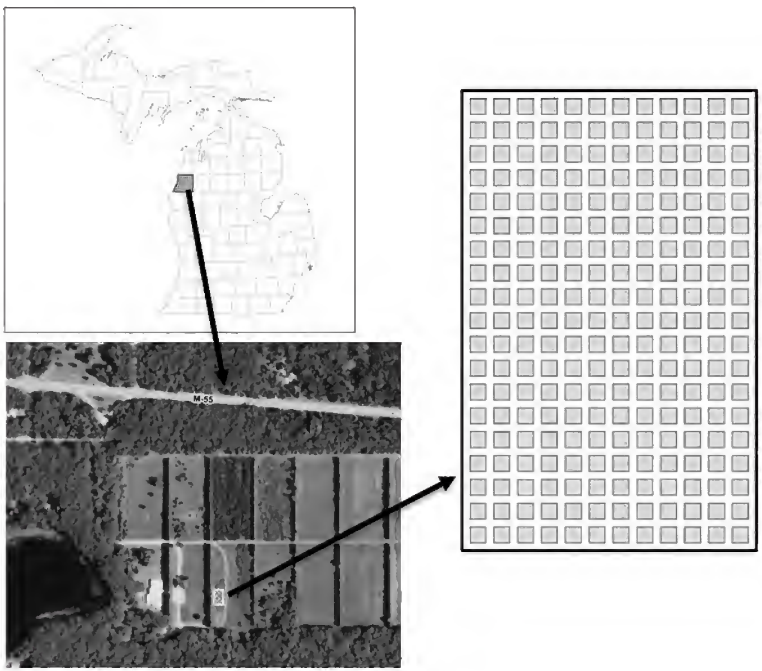

FIGURE 1. Chittenden Nursery, Manistee County, Michigan. Location of the approximately 500 m^2 rectangular study site within the nursery and a diagram showing the layout of the 228 1 m^2 study plots separated by 0.5 m buffers.

chosen to be planted based on their specific guild and historic presence in Michigan sand prairies. Three species per guild were selected. Selection criteria reflect a balance of conservative versus less-conservative native species. Species were further selected to include a variety of ecological characteristics, including both late vs. early season flowering species, both C4 and C3 photosynthetic pathways (among the grasses), and, with respect to soil nutrient relationships, both nitrogen fixers (legumes) and nitrogen extractors. The plants selected for seeding were placed into one of five groups—legumes, early season forbs, late season forbs, grasses, and background species. The species in each of these five groups are listed in Table 1. Twelve of the treatments consisted of one of the first three functional groups (legumes, early season forbs, late season forbs), grasses, and background species, for a maximum of 13 species per treatment. Background species were selected and added to the seed mixes in order to mimic a more traditional species rich and diverse seed mixture similar to those used in sand prairie restoration activities.

Each species other than the background species was seeded at either a high concentration (10,000 seeds/m^2) or a low concentration (1,000 seeds/m^2). The background species were seeded at a density of 500 seeds/m^2. Seeding densities were equally divided for each species so that an equal number of seeds per species were represented in the mix. Seeds were obtained from Michigan Wildflower Farm, Portland, Michigan. All seeds are of local Michigan genotypes.

In January 2009, seeds were counted and weighed to determine the number of seeds per gram for each species. The appropriate number of seeds were mixed with clean, moist sand and placed in a freezer for a minimum of two weeks to simulate cold-moist stratification before being sown into the field. In March 2009, all seed mixes, except those containing legumes, were sown evenly by hand into treatment plots. Due to the unavailability of seed in the spring, legume seed treatments (including their grass and background species components) were sown in December 2009. All seeded group comparisons and control plots were randomly assigned. Each treatment was replicated 12 times for a

TABLE 1. Species in each of the five seeding groups and associated coefficient of conservatism (CC) values.

Seeding Group	Species	CC Value
Legumes	*Lupinus perennis* L.	7
	Desmodium canadense (L.) DC.	3
	Lespedeza capitata Michx.	5
Early Season Forbs	*Penstemon hirsutus* (L.) Willd.	5
	Asclepias tuberosa L.	5
	Anemone virginiana L.	3
Late Season Forbs	*Symphyotrichum laeve* (L.) Á. Löve & D. Löve	5
	Solidago nemoralis Aiton	2
	Solidago speciosa Nutt.	5
Grasses	*Andropogon gerardii* Vitman	5
	Schizachyrium scoparium (Michx.) Nash	5
	Sorghastrum nutans (L.) Nash	6
Background Species	*Coreopsis lanceolata* L.	8
	Euphorbia corollata L.	4
	Liatris aspera Michx.	4
	Monarda fistulosa L.	2
	Oenothera biennis L.	2
	Rudbeckia hirta L.	1
	Verbena stricta Vent.	4

total of 228 plots. The composition by functional group of each of the seeding treatments is given in Table 2. Treatments that are listed as having 24 replicates in Table 2 are those that received separate March and December seedings of 12 replicates each.

Baseline Data Collection

In July 2009, before the growth of any of the seed treatments, visual estimates were made of the percentage cover of vegetation for each plot. Measurements of the percentage of vegetative cover were broken down into seven different groupings in order to further explore the differences in each treatment group. These groupings were: (i) all species encountered ("total"), (ii) all non-native species encountered ("non-native"), (iii) all native species encountered ("native"), (iv) all non-seeded species encountered, both native and non-native ("resident"), (v) all seeded species encountered ("planted"), (vi) all grass species encountered ("grass"), and (vii) all forb species encountered ("forb").

Initial estimates of percentage cover for each plot were averaged across the entire 228 plot study site. Mean species richness and diversity values were derived from the percentage cover data of each plot and were averaged across the entire study site. Baseline measurement data, representative of pre-restoration conditions, are provided in Table 3.

Floristic quality index (FQI) values of the different seed mixes were calculated (Table 4). These values may provide insight on the extent that restoration plot FQI may be predetermined by seed mix design. Species nomenclature follows The Taxonomic Name Resolution Service (2014), which is an online tool for automated standardization of plant names as described by Boyle et al. (2013).

In late August 2009, a floristic quality assessment (FQA) was completed for the one-half hectare area that surrounds the study site. This area was not affected by the site-preparation herbicide treatments. The surrounding area included a mix of disturbed, developed land, mesic-mixed forest, and xeric open fields. This FQA provides baseline data for species that occur in the immediate area adjacent to the study site and may provide information for future measures (e.g., seed bank germination

TABLE 2. Composition by functional groups of the 15 treatment groups. N indicates the number of replicates for each treatment. In addition to these treatments, there were 12 control plots that received no treatment.

Treatment Group	N
Early Season Flowering Forbs	
1. Early Season Forbs (high); Grasses (low); Background Species	12
2. Early Season Forbs (low); Grasses (high); Background Species	12
3. Early Season Forbs (low); Grasses (low); Background Species	12
4. Early Season Forbs (low); Background Species	12
Late Season Flowering Forbs	
5. Late Season Forbs (high); Grasses (low); Background Species	12
6. Late Season Forbs (low); Grasses (high); Background Species	12
7. Late Season Forbs (low); Grasses (low); Background Species	12
8. Late Season Forbs (low); Background Species	12
Legumes	
9. Legumes (high); Grasses (low); Background Species	12
10. Legumes (low); Grasses (high); Background Species	12
11. Legumes (low); Grasses (low); Background Species	12
12. Legumes (low); Background Species	12
Grasses and Background Species Only	
13. Grasses (high); Background Species	24
14. Grasses (low); Background Species	24
Background Species Only	
15. Background Species	24

TABLE 3. Mean pre-restoration, baseline data collection values for percentage cover, species richness, FQI, and H' of the entire 228-plot study site in July 2009.

Variable	Value
Cover (%)	
Native	42.22
Non-Native	57.78
Resident	99.97
Planted	0.03
Grass	0.75
Forb	99.25
Total	100.00
Species Richness	
Total	6.98
Native	2.08
Non-Native	4.90
Resident	6.92
Planted	0.06
Grass	0.37
Forb	6.61
Diversity	
Floristic Quality Index (FQI)	0.99
Shannon Diversity Index (H')	1.11

and colonization). Plants that could not be readily identified were trimmed with scissors at the base and placed into plastic bags. The bags were labeled with key plant and environmental characteristics (e.g. soil conditions, exposure to shade/sun). The plants were transported on ice back to Grand Valley State University in Allendale, Michigan, where they could further be identified. The list of species collected is provided in Table 5. It is important to note that only species that were identifiable in the fall were included in the list. Many native and non-native plants, especially early season forbs and grasses, may have been excluded in this initial sampling due to their lack of key reproductive structures.

Data Collection

Data on the percentage cover of each species and on species richness were collected from the study plots in mid-July each year from 2009 through

TABLE 4. Floristic quality index (FQI) of each of the seed mixes.

Seed Mix	FQI
Early Season Flowering Forbs	
Early Season Flowering Forbs; Grasses; Background Species	14.98
Early Season Flowering Forbs; Background Species	12.02
Late Season Flowering Forbs	
Late Season Flowering Forbs; Grasses; Background Species	14.70
Late Season Flowering Forbs; Background Species	11.70
Legumes	
Legumes; Grasses; Background Species	15.53
Legumes; Background Species	12.65
Grasses and Background Species Only	
Grasses; Background Species	12.97
Background Species Only	
Background Species	9.45
Entire Seeded Site	18.58

2011. These data were used to calculate the floristic quality index (FQI) and the Shannon diversity index (*H'*) for each plot. If the species identification of any plant was not possible due to the plant's early life stage, it was flagged for later identification. Plants that were ultimately unidentified were not incorporated into the data analysis of percentage cover.

The percentage cover of each species, of bare-ground, and of litter percent were visually estimated in 2009 and 2010. In order to provide more objective quantitative data, percentage cover was determined in 2011 using the point-intercept method. This method involves dropping a metal pin at 50 points along an evenly spaced grid that covered the entire plot. Any vegetative part of a plant that touched the pin was counted as an occurrence of that plant at that particular point. Any piece of litter or bare-ground that touched the pin was also accounted for. The total number of occurrences of an individual species in the plot was then divided by the total number of occurrences of all species in the plot to provide the relative percentage cover of that species. Any species noted visually as present in a plot that was not encountered by the point-intercept method was marked as being a trace amount occurrence (0.0625% cover) of the plot. In 2009, only the 144 non-legume plots were evaluated. In 2010 and 2011, all 228 plots were evaluated. In an effort to maintain consistent data collection efforts, estimates of percentage cover were calibrated between the same two researchers throughout the course of the multi-year study. This was a necessary approach, as cover estimates are subjective between individuals.

Species richness was calculated by counting the number of species that occurred in each plot. Those species for which identification was not possible due to the plant's early life stage were included as part of these calculations if they were identifiable to the extent that they could be determined to be different from species otherwise noted in the plot. A list was kept of all species identified, and all unknowns were described in detail.

Floristic quality index (FQI) measures the overall quality of an area (Swink and Wilhelm 1994) based on coefficient of conservatism (CC) values provided by the Michigan DNR (Hermann et al. 1996). Assigning CC values to individual plant species provides an approach to ranking the quality of plant communities (Swink and Wilhelm 1994). These values range from 0 to 10 and are assigned only to native species. A plant with a low value would represent a species that is common, can persist in highly degraded areas, and is likely not indicative of a high quality remnant community, for example, *Solidago canadensis* L., for which CC = 0, and *Rudbeckia hirta* L., for which CC=1. Conversely, a species with a high value represents a species that would likely be found only in a place indicative of an intact remnant of a natural ecosystem, for example, *Lithospermum canescens* (Michx.)

TABLE 5. List of species collected at the study site prior to the growth of any of the seed treatments. Non-native species are indicated by an asterisk (*). The coefficient of conservatism (CC) value for the native species are taken from Hermann et al. (1996).

Species	CC Value	Species	CC Value
Acer rubrum L.	1	*Plantago rugelii* Decne.	0
Ambrosia artemisiifolia L.	0	*Populus tremuloides* Michx.	1
Andropogon gerardii Vitman	5	*Potentilla argentea* L.*	-
Asclepias syriaca L.	1	*Potentilla recta* L.*	-
Bromus inermis Leyss.*	-	*Pseudognaphalium obtusifolium* (L.) Hilliard & B.L. Burtt	2
Centaurea stoebe L.*	-	*Pteridium aquilinum* (L.) Kuhn	0
Cichorium intybus L.*	-	*Robinia viscosa* Vent.*	-
*Cirsium vulgare** (Savi) Ten.*	-	*Rudbeckia hirta* L.	1
Clinopodium vulgare L.	3	*Sassafras albidum* (Nutt.) Nees	5
Comptonia peregrina (L.) J.M. Coult.	6	*Schizachyrium scoparium* (Michx.) Nash	5
Conyza canadensis (L.) Cronquist	0	*Solidago canadensis* L.	1
Daucus carota L.*	-	*Sorghastrum nutans* (L.) Nash	6
Elaeagnus umbellata Thunb.*	-	*Symphyotrichum laeve* (L.) Á. Löve & D. Löve	5
Elymus repens (L.) Gould.*	-	*Symphyotrichum ontarionis* (Wiegand) G.L. Nesom	6
Erigeron strigosus Muhl. Ex Willd.	4	*Tragopogon pratensis* L.*	-
Fragaria virginiana Mill.	2	*Trifolium arvense* L.*	-
Holosteum umbellatum L.*	-	*Triosteum perfoliatum* L.	5
Hypericum perforatum L.*	-	*Verbascum blattaria* L.*	-
Juniperus virginiana L.	3	*Verbascum thapsus* L.*	-
Lepidium virginicum L.	0	*Verbena hastata* L.	4
Leucanthemum vulgare Lam.*	-	*Vitis aestivalis* Michx.	6
Lupinus perennis L.	7		
Melilotus alba (L.) Lam.*	-		
Melilotus officinalis (L.) Lam.*	-		
Monarda fistulosa L.	2		
Monarda punctata L.	4		
Oenothera biennis L.	2		
Panicum capillare L.	1		
Pinus strobus L.	3		

Lehm.), for which CC=10, and *Ceanothus americanus* L., for which CC = 9 (Swink and Wilhelm 1994, Hermann et al. 1996). FQI is calculated by Equation 1, where C is the average of CC values from native plants species in the sampled community, and N is the total number of native species in the community. Non-native plant species are not included in this calculation.

Equation 1: $FQI = C\sqrt{N}$

According to Hermann et al. (1996), an FQI greater than 50 indicates an area of high conservatism, and an area with an FQI greater than 35 is considered to be floristically high quality in Michigan.

Diversity was expressed by the Shannon diversity index (H') (Shannon 1948), which equals (–1) times the sum for all native species in the sampled community of the product of the relative cover (p_i) of each species and its natural logarithm, as shown in Equation 2.

$$H' = -\sum_{i=1}^{N} p_i \ln p_i$$

Statistical Analysis

Measurements of the percentage of vegetative cover and of species richness were broken down into the previously mentioned seven different groupings: total, non-native, native, resident, planted, grass, and forb groupings.

Treatments were compared by both forb and grass groupings for percentage cover, species rich-

ness, and diversity. Grass groupings include control plots, background species only (treatment 15 in Table 2), forbs, including legumes with no grass (treatments 4, 8, and 12), high grass (treatments 2, 6, 10, 13), and low grass (treatments 1, 3, 5, 7, 9, 11, and 14). Forb groupings include control plots, background forb species only (treatments 13–15), early season flowering forbs (treatments 1-3), and late season flowering forbs (treatments 5–7), legumes (treatments 9-11), and forb treatments containing either early season flowering forbs, late season flowering forbs, or legumes that contained no grass (treatments 4, 8, and 12).

Exploratory analysis for percentage cover of bare-ground and litter values revealed that they were normally distributed. In order to produce normality in the other variables, a variety of transformations were used. The percentage cover values for the native, planted, and grass groupings were square root transformed in order to correct for their positive skew. The percentage cover values for the non-native, resident, and forb groupings were transformed by formula, *Normal Value* $(X) = \sqrt{K - X}$. (where K is a constant from which each score is subtracted so that the smallest score is 1), to correct for their negative skew (Tabachnick and Fidell 2007). Once the data were normalized, a one-way analysis of variance (ANOVA) was used to test whether there were significant differences in percentage cover between treatment groups in 2011. In order to further compare the differences among multiple comparison groups, Tukey post-hoc tests were run between all treatment groups. Comparisons were considered significantly different at $p \leq 0.05$ after taking into account the Bonferroni correction. All statistical analyses were performed using PASW 18 (SPSS Inc. 2011). Data are reported as non-transformed values.

Exploratory analysis for species richness and diversity values revealed that they were normally distributed. A one-way ANOVA was used to test whether there were significant differences among treatment groups in species richness and diversity in 2011. In order to further compare the differences among multiple comparison groups, Tukey post-hoc tests were run between all treatment groups. Comparisons were considered significantly different at $p \leq 0.05$ after taking into account the Bonferroni correction. All statistical analyses were performed using PASW 18 (SPSS Inc. 2011).

An independent sample T-test was used to determine if there were significant differences in measured variables between spring seeded treatments and fall seeded legume treatments. All treatments were statistically similar.

RESULTS

Baseline Floristic Quality Assessment

A total of 50 different plant species were encountered during the baseline floristic quality assessment of the surrounding areas of the study site in 2009, including 31 native and 19 non-native species (Table 5). The average coefficient of conservatism (CC) value was 2.94, resulting in a calculated floristic quality index (FQI) of 16.37, which indicates that the area was not considered a floristically high quality site (Hermann et al. 1996). A list of resident species encountered within the study plots by year is provided in Table 6.

Baseline vegetative cover data indicate that the most prevalent species were *Conyza canadensis* (L.) Cronquist (mean = 41%), *Centaurea stoebe* L. (mean = 23%), and *Potentilla argentea* L. (mean = 22%). The most prevalent grass species were *Bromus inermis* Leyss. (mean = 1%) and *Agrostis hyemalis* (Walter) Britton, Sterns & Poggenb. (mean = 1%).

Comparison among Grass Treatment Groups after Three Years

Percentage Cover

The data reported in this section are summarized in Table 7. Significant differences ($p<0.050$) were found among treatment groups in the percentage cover

TABLE 6. Resident species encountered within the study site in each of the years of the study. Non-native species are indicated by an asterisk (*).

2009	2010	2011
Acer rubrum L.	*Acer rubrum* L.	*Asclepias syriaca* L.
Agrostis hyemalis (Walter) Brit., Sterns & Pogg.	*Agrostis hyemalis* (Walter) Brit., Sterns & Pogg.	*Acer rubrum* L.
Arenaria serpyllifolia L.*	*Arenaria serpyllifolia* L.*	*Agrostis hyemalis* (Walter) Brit., Sterns & Pogg.
Asclepias syriaca L.	*Asclepias syriaca* L.	*Arenaria serpyllifolia* L.*
Berteroa incana (L.) DC.*	*Berteroa incana* (L.) DC.*	*Berteroa incana* (L.) DC.*
Bromus inermis Leyss.*	*Bromus inermis* Leyss.*	*Bromus inermis* Leyss.*
Centaurea stoebe L.*	*Centaurea stoebe* L.*	*Centaurea stoebe* L.*
Clinopodium vulgare L.	*Clinopodium vulgare* L.	*Clinopodium vulgare* L.
Conyza canadensis (L.) Cronquist	*Conyza canadensis* (L.) Cronquist	*Conyza canadensis* (L.) Cronquist
Danthonia spicata (L.) Roem. & Schult.	*Danthonia spicata* (L.) Roem. & Schult.	*Danthonia spicata* (L.) Roem. & Schult.
Elymus repens (L.) Gould*	*Elymus repens* (L.) Gould*	*Elymus repens* (L.) Gould*
Erigeron annuus (L.) Desf.	*Erigeron annuus* (L.) Desf.	*Fragaria virginiana* Mill.
Fragaria virginiana Mill.	*Fragaria virginiana* Mill.	*Galium aparine* L.
Gnaphalium obtusifolium L.	*Gnaphalium obtusifolium* L.	*Gnaphalium obtusifolium* L.
Hieracium auranticum L.*	*Hieracium auranticum* L.*	*Hieracium auranticum* L.*
Hieracium caespitosum Dumort.*	*Hieracium caespitosum* Dumort.*	*Hieracium caespitosum* Dumort.*
Hypericum perforatum L.*	*Hypericum perforatum* L.*	*Hypericum perforatum* L.*
Lepidium virginicum L.	*Lepidium virginicum* L.	*Lepidium virginicum* L.
Leucanthemum vulgare Lam.*	*Leucanthemum vulgare* Lam.*	*Leucanthemum vulgare* Lam.*
Oxalis stricta L.	*Oxalis stricta* L.	*Monarda punctata* L.
Panicum capillare L.	*Panicum capillare* L.	*Oxalis stricta* L.
Physalis heterophylla Nees	*Physalis heterophylla* Nees	*Panicum capillare* L.
Poa compressa L.*	*Poa compressa* L.*	*Physalis heterophylla* Nees
Potentilla argentea L.*	*Potentilla argentea* L.*	*Poa compressa* L.*
Potentilla recta L.*	*Potentilla recta* L.*	*Potentilla argentea* L.*
Prunus serotina Ehrh.	*Prunus serotina* Ehrh.	*Potentilla recta* L.*
Quercus rubra L.	*Quercus rubra* L.	*Prunus serotina* Ehrh.
Rubus occidentalis L.	*Rubus occidentalis* L.	*Quercus rubra* L.
Rumex acetosella L.*	*Rumex acetosella* L.*	*Rubus occidentalis* L.
Taraxacum officinale F.H. Wigg.*	*Taraxacum officinale* F.H. Wigg.*	*Rumex acetosella* L.*
Tragopogon pratensis L.*	*Tragopogon pratensis* L.*	*Taraxacum officinale* F.H. Wigg.*
Trifolium arvense L.*	*Trifolium arvense* L.*	*Tragopogon pratensis* L.*
Trifolium pratense L.*	*Trifolium pratense* L.*	*Trifolium arvense* L.*
Trifolium repens L.*	*Trifolium repens* L.*	*Trifolium repens* L.*
Verbascum thapsus L.*	*Verbascum thapsus* L.*	*Verbascum thapsus* L.*
Veronica arvensis L.*	*Veronica arvensis* L.*	

of the bare-ground ($F_{4,227}$=5.79, p<0.001), native ($F_{4,227}$=11.20, p<0.001), non-native ($F_{4,227}$=11.41, p<0.001), resident ($F_{4,227}$=24.09, p<0.001), planted ($F_{4,227}$=23.90, p<0.001), grass ($F_{4,227}$=30.39, p=0.014), and forb ($F_{4,227}$=30.39, p=0.014) groupings. There were no significant differences (p>0.050) among treatment groups in litter cover ($F_{4,227}$=0.70, p=0.595, mean = 17.86%).

Bare-ground cover averaged 36.03% across all grass treatment groups. High grass treatments had significantly lower bare-ground cover (mean = 31.85%, p<0.050) than all other treatment groups except the control plots (mean = 37.20%), which showed high variability.

Native cover averaged 19.34% across all grass treatment groups. High grass treatments had significantly higher native cover (mean = 35.02%, p<0.050) than all other treatment groups. Low grass treatments had significantly higher native cover (mean = 22.99%, p<0.050) than all other treatment groups except the high grass treatments. There were no significant differences in native cover among all other treatment groups (mean range: 8.01–16.94%) Comparison of mean native cover in each of these treatment groups is shown in Figure 2.

Non-native cover averaged 80.54% across all grass treatment groups. High grass treatments had significantly lower non-native cover (mean = 64.38%, p<0.050) than all other treatment groups, while low grass treatments had significantly lower cover (mean = 77.01%, p<0.050) than all other treatment groups except the high grass treatment. There were no significant differences in non-na-

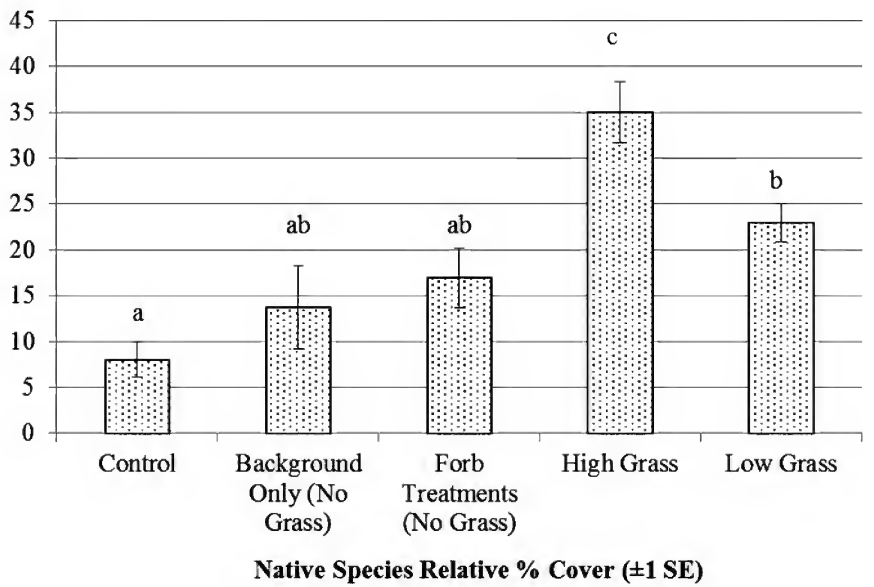

FIGURE 2. Grass Treatment Comparison: Mean relative percentage cover of species in the native grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different (p≤.05) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

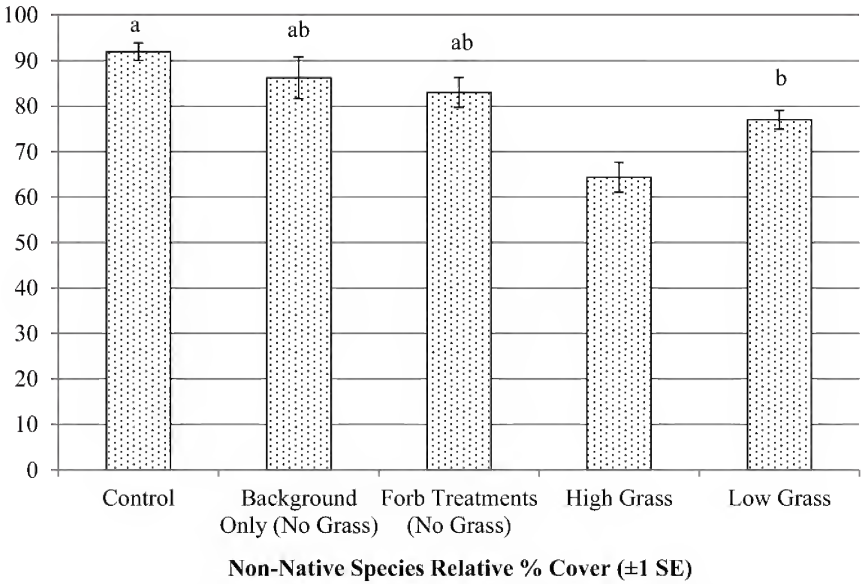

FIGURE 3. Grass Treatment Comparison: Mean relative percentage cover of species in the Non-Native grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

tive cover among all other treatment groups (mean range: 83.06–91.99%). Comparison of mean non-native cover in each of these treatment groups is shown in Figure 3.

Resident cover averaged 86.45% across all grass treatment groups. Resident cover in control plots (mean = 99.83%) was significantly greater ($p<0.050$) than all other treatments (mean range: 68.00–91.12%). High grass treatments had significantly lower resident cover (mean = 68.00%, $p<0.050$) than all other treatments. Low grass treatments had significantly lower (mean = 82.71%, $p<0.050$) resident cover than the treatment with background only species and no grass treatment and the control plots.

Planted cover averaged 13.43% across all grass treatment groups. Control plots had significantly less planted cover (mean = 0.17%, $p<0.050$) than all other treatment groups (mean range: 8.88–31.40%). High grass treatments had significantly higher planted cover (mean = 31.40%, $p<0.050$) than all other treatment groups. Low grass treatments had significantly higher planted cover (mean = 17.29%, $p<0.050$) than the background only treatments with no grass treatments (mean = 8.88%), but was not significantly different ($p>0.050$) from the forb treatments with no grass (mean = 9.43%).

Grass cover averaged 10.35% across all grass treatment groups. High grass treatments had significantly more grass cover (mean = 28.47%, $p<0.050$) than all other treatment groups. Low grass treatments had significantly higher grass

TABLE 7. Mean percentage cover values of bare-ground and litter and of the native, non-native, resident, planted, grass, and forb groupings in the grass treatment groups in 2011. Percentage cover in each grouping was tested only against the treatments within the same grouping. Values within a column that do not share a superscript letter are significantly different ($p \leq 0.05$) as determined by Tukey post-hoc analysis.

Treatment Group	Bare-ground	Litter	Native	Non-Native	Resident	Planted	Grass	Forb
Control Plots	$37.20^{ab}(\pm1.56)$	$18.45(\pm1.66)$	$8.01^{a}(\pm1.93)$	$91.99^{a}(\pm1.93)$	$99.83(\pm0.17)$	$0.17(\pm0.17)$	$3.32(\pm1.22)$	$96.68(\pm1.22)$
Background Only, No Grass	$37.04^{b}(\pm1.62)$	$16.93(\pm1.35)$	$13.73^{ab}(\pm4.59)$	$86.27^{ab}(\pm4.59)$	$91.12^{b}(\pm4.25)$	$8.88^{b}(\pm4.25)$	$3.88(\pm2.35)$	$96.12(\pm2.35)$
Forb Treatment, No Grass	$38.38^{b}(\pm1.09)$	$18.17(\pm1.00)$	$16.94^{ab}(\pm3.29)$	$83.06^{ab}(\pm3.29)$	$90.57^{ab}(\pm2.03)$	$9.43^{ab}(\pm2.03)$	$4.93(\pm2.23)$	$95.07(\pm2.23)$
High Grass	$31.85^{a}(\pm1.20)$	$18.65(\pm0.51)$	$35.02(\pm3.28)$	$64.38(\pm3.28)$	$68.00^{c}(\pm3.17)$	$31.40^{c}(\pm3.17)$	$28.47^{b}(\pm3.01)$	$71.53^{a}(\pm3.01)$
Low Grass	$35.67^{b}(\pm0.64)$	$17.66(\pm0.50)$	$22.99^{b}(\pm2.06)$	$77.01^{b}(\pm2.06)$	$82.71^{a}(\pm1.92)$	$17.29^{a}(\pm1.92)$	$11.14^{a}(\pm1.40)$	$88.86^{b}(\pm1.40)$

TABLE 8. Mean species richness values of total, native, non-native, resident, planted, grass, and forb groupings in the grass treatment groups in 2011. Species richness in each grouping was tested only against the treatments within the same grouping. Values within a column that do not share a superscript letter are significantly different ($p \leq 0.05$) as determined by Tukey post-hoc analysis.

Treatment Group	Total	Native	Non-Native	Resident	Planted	Grass	Forb
Control Plots	$7.75^{b}(\pm0.54)$	$2.33(\pm0.28)$	$5.25(\pm0.49)$	$7.33(\pm0.47)$	$0.25(\pm0.13)$	$0.83(\pm0.24)$	$6.92(\pm0.36)$
Background Only, No Grass	$9.92^{bc}(\pm0.60)$	$4.63^{b}(\pm0.61)$	$5.25(\pm0.41)$	$7.08(\pm0.60)$	$2.79^{b}(\pm0.44)$	$0.75(\pm0.32)$	$9.17^{a}(\pm0.50)$
Forb Treatment, No Grass	$10.61^{c}(\pm0.49)$	$5.33^{b}(\pm0.42)$	$5.28(\pm0.23)$	$7.25(\pm0.29)$	$3.36^{b}(\pm0.29)$	$0.94(\pm0.19)$	$9.69^{a}(\pm0.38)$
High Grass	$12.88^{a}(\pm0.31)$	$7.48^{a}(\pm0.26)$	$5.37(\pm0.21)$	$7.25(\pm0.21)$	$5.60^{a}(\pm0.19)$	$3.50^{b}(\pm0.09)$	$9.40^{a}(\pm0.28)$
Low Grass	$12.49^{a}(\pm0.29)$	$6.96^{a}(\pm0.25)$	$5.50(\pm0.15)$	$7.43(\pm0.20)$	$5.03^{a}(\pm0.19)$	$2.63^{a}(\pm0.13)$	$9.86^{a}(\pm0.23)$

cover (mean = 11.14%, $p<0.050$) than all other treatment groups except the high grass treatments.

Forb cover averaged 89.65% across all grass treatment groups. High grass treatments had significantly less forb cover (mean = 71.53%, $p<0.050$) than all other treatment groups. Low grass treatments had significantly less forb cover (mean = 88.86%, $p<0.050$) than all other treatment groups except the high grass treatments.

Species Richness

The data reported in this section are summarized in Table 8. There were significant differences ($p<0.050$) in species richness among treatment groups in the total ($F_{4,227}=16.09$, $p<0.001$), native ($F_{4,227}=19.78$, $p<0.001$), planted ($F_{4,227}=40.16$, $p<0.001$), grass ($F_{4,227}=51.68$, $p<0.001$), and forb ($F_{4,227}=5.32$, $p<0.001$) groupings. There were no significant differences ($p>0.050$) in species richness among grass treatment groups in the non-native ($F_{4,227}=0.26$, $p=0.905$, mean = 5.33 species) and resident ($F_{4,227}=0.21$, $p=0.932$, mean = 7.27 species) groupings.

The total species richness averaged 10.73 species across all grass treatment groups. It averaged significantly lower for the control plots (mean = 7.75 species, $p<0.050$) than for all forb treatments with no grass (mean = 10.61 species), high grass treatments (mean = 12.88 species), and low grass treatments (mean = 12.49 species). High grass and low grass treatments exhibited significantly higher total species richness than all other grass treatment groups ($p<0.050$). Comparison of mean total species richness in each of these treatment groups is shown in Figure 4.

Species richness of the native grouping averaged 5.35 species across all grass treatment groups. The control plots had significantly lower native species richness (mean = 2.33 species, $p<0.050$) than all other treatment groups (mean range: 4.63–7.48 species). High grass and low grass treatments had significantly higher native species richness (mean = 7.48 and 6.96 species, respectively, $p<0.050$) than all other treatment groups. Comparison of mean native species richness in each of these treatment groups is shown in Figure 5.

Species richness of the planted grouping averaged 3.41 species across all grass treatment groups. All treatment groups had significantly higher planted species richness (mean range: 2.79–5.60 species, $p<0.050$) than the control plots (mean = 0.25 species). High grass and low grass treatments had significantly higher planted species richness (mean = 5.60 and 5.03 species, respectively, $p<0.050$) than all other treatment groups.

Species richness of the grass grouping averaged 1.73 species across all grass treatment groups. High grass treatments had significantly higher grass species richness (mean = 3.50 species, $p<0.050$) than all other treatments. Low grass treatments had significantly higher grass species richness (mean = 2.63 species, $p<0.050$) than all other treatments.

Species richness of the forb groupings averaged 9.01 species across all grass treatment groups. All treatment groups had significantly higher forb species richness (mean range: 9.17–9.86 species, $p<0.050$) than the control plots (mean = 6.92 species).

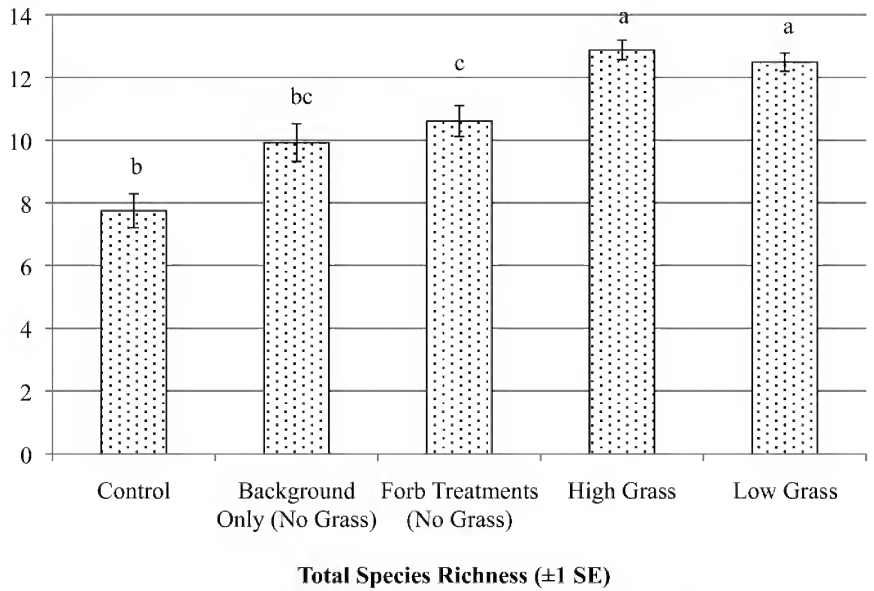

FIGURE 4. Grass Treatment Comparison: Mean species richness encountered in the Total grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

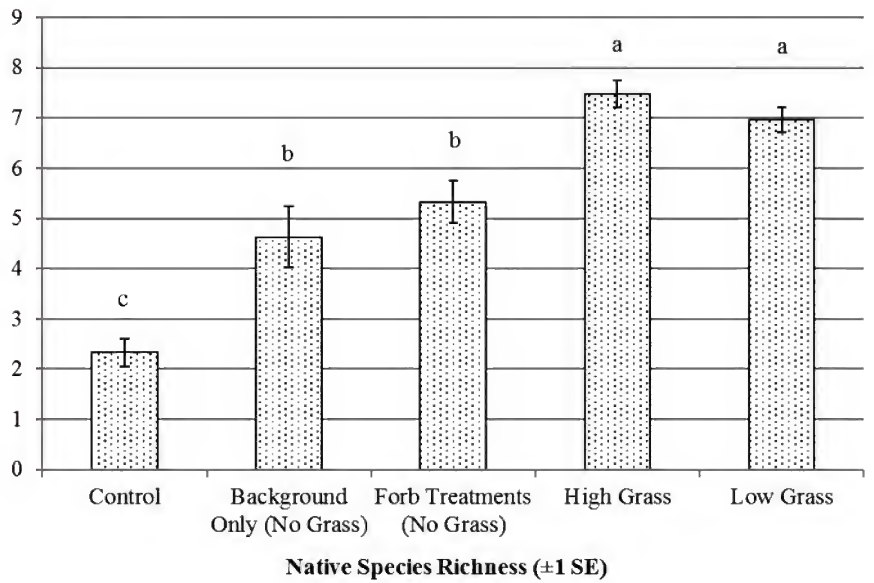

FIGURE 5. Grass Treatment Comparison: Mean species richness encountered in the Native grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

TABLE 9. Comparison of the mean floristic quality index (FQI) and Shannon diversity index (H') values of the grass treatment groups and selected non-grass treatment groups in 2011. Each different measure was tested independently. Values within a column that do not share a superscript letter are significantly different (p≤0.05) as determined by Tukey post-hoc analysis.

Treatment Group	FQI	H′
Control Plots	1.37(±0.56)	1.26^{b}(±0.09)
Background Only, No Grass	6.56^{a}(±0.45)	1.40^{ab}(±0.06)
Forb Treatment, No Grass	6.71^{a}(±0.39)	1.34^{b}(±0.06)
High Grass	10.37^{c}(±0.17)	1.61^{a}(±0.04)
Low Grass	9.35^{b}(±0.18)	1.49^{ab}(±0.04)

Floristic Quality and Diversity

There were significant differences among treatment groups in FQI values ($F_{4,227}$=87.83, p<0.001). The mean FQI of grass treatment groups was 6.87. The FQI of the control plots in 2011 was significantly lower (mean = 1.37) than that of all other treatments (mean range: 6.56–10.37). The high grass treatments had a significantly higher FQI (mean = 10.37) than all other treatment groups while low grass treatments had a significantly higher FQI (mean = 9.35) than all other treatments except the high grass treatments (Table 9; Figure 6).

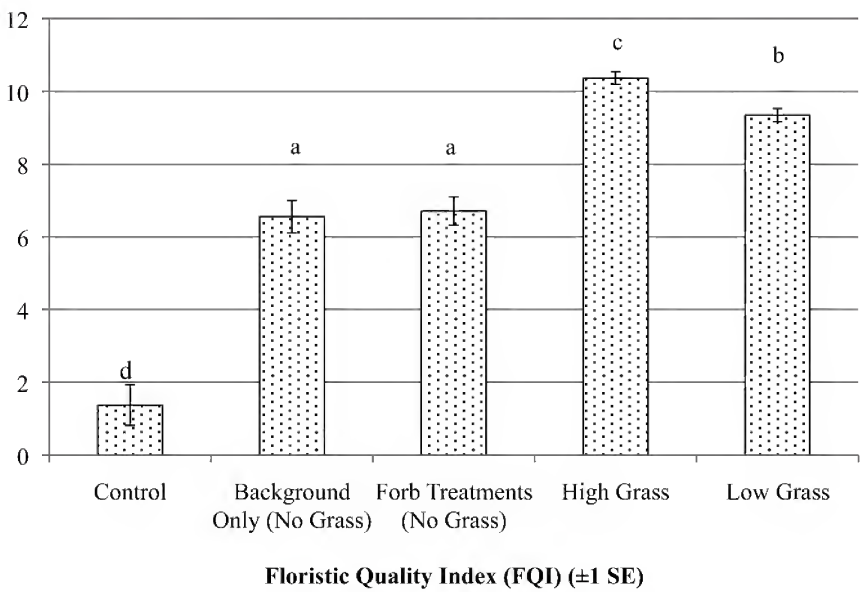

FIGURE 6. Grass Treatment Comparison: Mean FQI for several treatment groups in 2011. Bars that do not share the same letter are significantly different (p≤.05) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

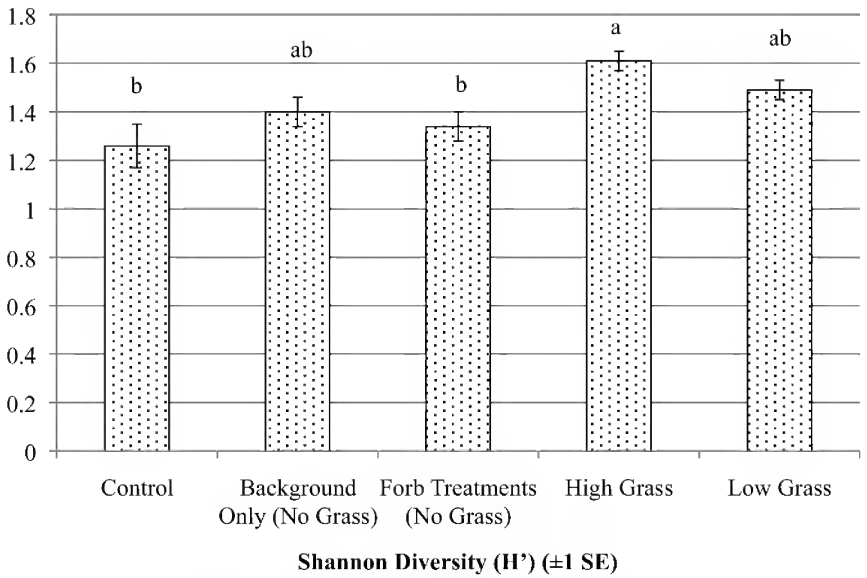

FIGURE 7. Grass Treatment Comparison: Mean H' for several treatment groups in 2011. Bars that do not share the same letter are significantly different (p≤.05) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

There were also significant differences among treatment groups in the Shannon diversity index (H') ($F_{4,227}$=4.43, p=0.002) (Table 9). The mean H' of the grass treatment groups was 1.42. The mean H′ of the high grass treatments was significantly higher (mean = 1.61) than for all other treatments except the background species only with no grass treatment (mean = 1.40) and the low grass treatment (mean = 1.49) (Table 9; Figure 7).

Comparison among Forb Treatment Groups after Three Years

Percentage Cover

The data reported in this section are summarized in Table 10. Significant differences (p<0.050) were found among treatment groups in the percentage cover of the bare-ground ($F_{4,227}$=2.99, p=0.020), native ($F_{4,227}$=8.75, p<0.001), non-native ($F_{4,227}$=8.33, p<0.001), resident ($F_{4,227}$=17.72, p<0.001), planted ($F_{4,227}$=18.51, p<0.001), grass ($F_{4,227}$=3.21, p=0.014), and forb ($F_{4,227}$=3.21, p=0.014) groupings. There were no significant differences (p>0.050) among treatment groups in litter cover ($F_{4,227}$=0.16, p=0.959, mean = 18.04%).

Bare-ground cover averaged 35.59% among forb treatment groups. The late season flowering forb treatment had significantly less (p<0.050) bare-ground cover than the legume treatments (mean = 33.40% and 37.96%, respectively).

Native cover averaged 21.24% among forb treatment groups. Treatments of early season flowering forbs and late season flowering forbs showed significantly

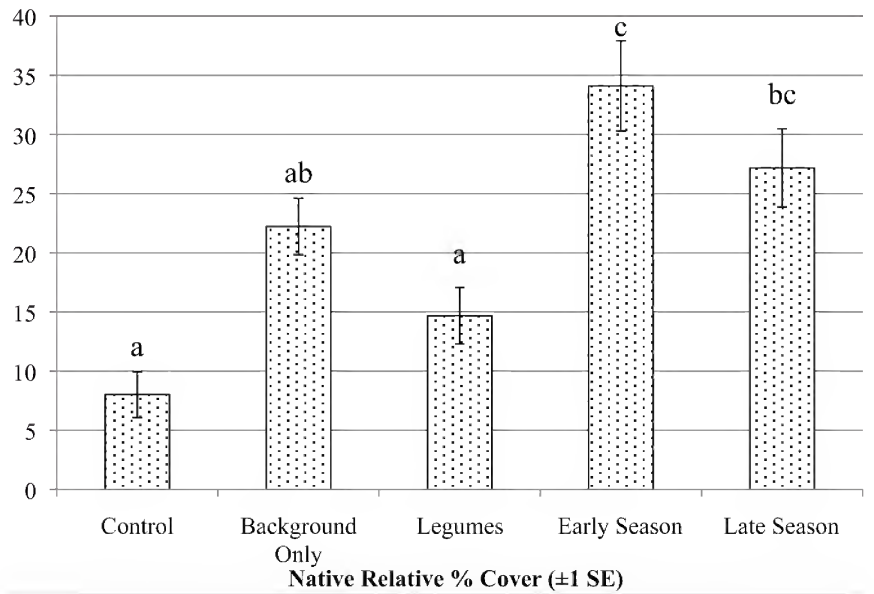

FIGURE 8. Forb Treatment Comparison: Mean relative percentage cover of species in the Native grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

higher percent native cover (mean = 34.10% and 27.19%, respectively, $p<0.050$) than the control plots (mean = 8.01%) and legume treatments (mean = 14.68%). Comparison of mean native cover in each of these treatment groups is shown in Figure 8.

Non-native cover averaged 78.61% among forb treatment groups. The control plots and legume treatments showed significantly higher non-native cover (mean = 91.99% and 84.57%, respectively, $p<0.050$) than early season and late season flowering forb treatments (mean = 65.90%, 72.81%, respectively). Comparison of mean non-native cover in each of these treatment groups is shown in Figure 9.

Resident cover averaged 84.39% among forb treatment groups. Resident cover in control plots (mean = 99.83%) was significantly greater ($p<0.050$) than in all other treatments (mean range: 71.16%–90.96%). Early season flowering forbs and late season flowering forbs (mean = 71.16% and 76.46%, respectively) had significantly lower ($p<0.050$) resident cover than all other treatments.

Planted cover averaged 15.46% among forb treatment groups. Control plots had significantly less planted cover (mean = 0.17%, $p<0.050$) than all other treatment groups (mean range: 8.29%–28.84%). Early season flowering forb and late season flowering forb treatments (mean = 28.84% and 23.54%, respectively) had significantly more ($p<0.050$) planted cover than all other treatment groups.

Grass cover averaged 11.93% among forb treatment groups. All treatment groups had significantly higher grass relative cover (mean range:

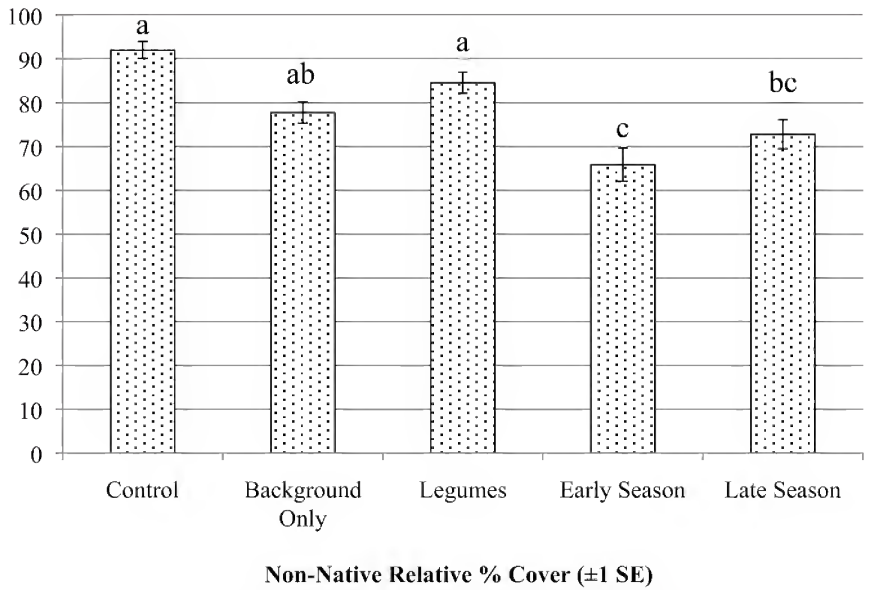

FIGURE 9. Forb Treatment Comparison: Mean relative percentage cover of species in the Non-Native grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different (p≤.05) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

8.30%–17.48%, $p<0.050$) than the control plots (mean = 3.32%). Early season flowering forbs and late season flowering forbs had significantly higher grass cover (mean = 17.48% and 16.18%, respectively, $p<0.050$) than all other treatment groups except the background only treatments (mean = 14.36%.

Forb cover averaged 88.07% among forb treatment groups. All treatment groups had significantly lower ($p<0.050$) forb (mean range: 82.52%–91.70%) relative cover than the control plots (mean = 96.68%). Early season flowering forbs and late season flowering forbs had significantly lower forb cover (mean = 82.52% and 83.82%, respectively) than all other treatment groups except the background only treatments (mean = 85.64%.

Species Richness

The data reported in this section are summarized in Table 11. There were significant differences ($p<0.050$) in species among treatment groups in the total ($F_{4,227}=9.99$, $p<0.001$), native ($F_{4,227}=15.77$, $p<0.001$), planted ($F_{4,227}=26.28$, $p<0.001$), grass ($F_{4,227}=3.98$, $p=0.004$), and forb ($F_{4,227}=9.96$, $p<0.001$) groupings. There were no significant differences ($p>0.050$) in species richness among forb treatment groups in the non-native ($F_{4,227}=0.65$, $p=0.629$, mean = 5.35 species) and resident ($F_{4,227}=0.38$, $p=0.825$, mean = 7.29 species) groupings.

The total species richness averaged 11.20 species among forb treatment groups. It averaged significantly less for the control plots (mean = 7.75 species,

TABLE 10. Mean percentage cover values of bare-ground and litter and of the native, non-native, resident, planted, grass, and forb groupings in the grass treatment groups in 2011. Percentage cover in each grouping was tested only against the treatments within the same grouping. Values within a column that do not share a superscript letter are significantly different ($p \leq 0.05$) as determined by Tukey post-hoc analysis.

Treatment Group	Bare-ground	Litter	Native	Non-Native	Resident	Planted	Grass	Forb
Control Plots	37.20^{ab}(±1.56)	18.45(±1.66)	8.01^{a}(±1.93)	91.99^{a}(±1.93)	99.83(±0.17)	0.17(±0.17)	3.32(±1.22)	96.68(±1.22)
Background Forbs Only	35.41^{ab}(±0.77)	17.97(±0.50)	22.24^{ab}(±2.39)	77.76^{ab}(±2.39)	83.55^{a}(±2.26)	16.45^{a}(±2.26)	14.36^{ab}(±2.08)	85.64^{ab}(±2.08)
Legumes	37.96^{a}(±1.01)	17.69(±0.80)	14.68^{a}(±2.39)	84.57^{a}(±2.39)	90.96^{a}(±1.58)	8.29^{a}(±1.58)	8.30^{a}(±1.80)	91.70^{a}(±1.80)
Early Season	33.99^{ab}(±1.22)	17.74(±0.68)	34.10^{c}(±3.80)	65.90^{c}(±3.80)	71.16^{b}(±3.65)	28.84^{b}(±3.65)	17.48^{bc}(±3.14)	82.52^{bc}(±3.14)
Late Season	33.40^{b}(±1.13)	18.33(±0.75)	27.19^{bc}(±3.31)	72.81^{bc}(±3.31)	76.46^{b}(±3.12)	23.54^{b}(±3.12)	16.18^{bc}(±3.15)	83.82^{bc}(±3.15)

TABLE 11. Mean species richness values of total, native, non-native, resident, planted, grass, and forb groupings in the forb treatment groups in 2011. Species richness in each grouping was tested only against the treatments within the same grouping. Values within a column that do not share a superscript letter are significantly different ($p \leq 0.05$) as determined by Tukey post-hoc analysis.

Treatment Group	Total	Native	Non-Native	Resident	Planted	Grass	Forb
Control Plots	7.75(±0.54)	2.33(±0.28)	5.25(±0.49)	7.33(±0.47)	0.25(±0.13)	0.83^{a}(±0.24)	6.92(±0.36)
Background Forbs Only	11.53^{a}(±0.29)	5.93^{a}(±0.25)	5.58(±0.17)	7.51(±0.21)	4.00^{a}(±0.19)	2.46^{b}(±0.19)	9.08^{b}(±0.21)
Legumes	11.83^{ab}(±0.49)	6.46^{a}(±0.43)	5.35(±0.23)	7.21(±0.29)	4.60^{a}(±0.32)	2.06^{ab}(±0.22)	9.77^{ab}(±0.38)
Early Season	13.25^{b}(±0.44)	8.06^{b}(±0.37)	5.15(±0.19)	7.29(±0.27)	5.92^{b}(±0.29)	2.56^{b}(±0.22)	10.71^{a}(±0.33)
Late Season	11.63^{a}(±0.35)	6.17^{a}(±0.30)	5.42(±0.23)	7.13(±0.26)	4.46^{a}(±0.20)	2.40^{b}(±0.19)	9.23^{b}(±0.26)

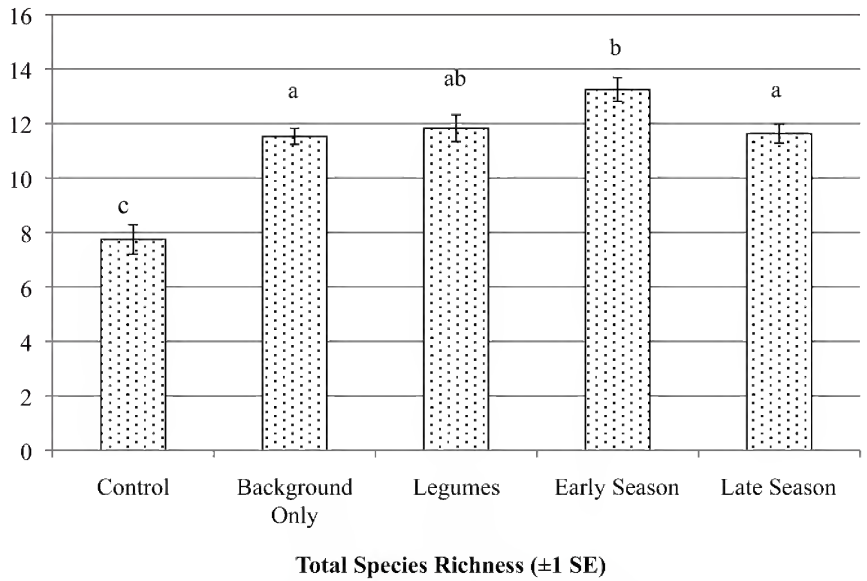

FIGURE 10. Forb Treatment Comparison: Mean species richness encountered in the Total grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

$p<0.050$) than for all other treatments. The early season flowering forb treatment had significantly higher total species richness (mean = 13.25 species, $p<0.050$) than all other treatments except the legumes (mean = 11.83 species). Comparison of mean total species richness in each of these treatment groups is shown in Figure 10.

Species richness of the native grouping averaged 5.79 species among forb treatment groups. The control plots had significantly lower native species richness (mean = 2.33 species, $p<0.050$) than all other treatment groups. The early season flowering forb treatment group had significantly higher native species richness (mean = 8.06 species, $p<0.050$) than all other treatment groups (mean range: 5.93–6.46 species). Comparison of mean native species richness in each of these treatment groups is shown in Figure 11.

Species richness of the planted grouping averaged 3.85 species among forb treatment groups. All treatment groups had significantly higher ($p<0.050$) planted species richness than the control plots (mean = 0.25 species). The early season flowering forbs had significantly greater planted species richness (mean = 5.92 species, $p<0.050$) than all other treatment groups (mean range: 4.00–4.60 species).

Species richness of the grass grouping averaged 2.06 species among forb treatment groups. All treatments except for legumes (mean = 2.06 species) exhibited significantly higher grass species richness (mean range: 2.40–2.56 species, $p<0.050$) than the control plots (mean = 0.83 species).

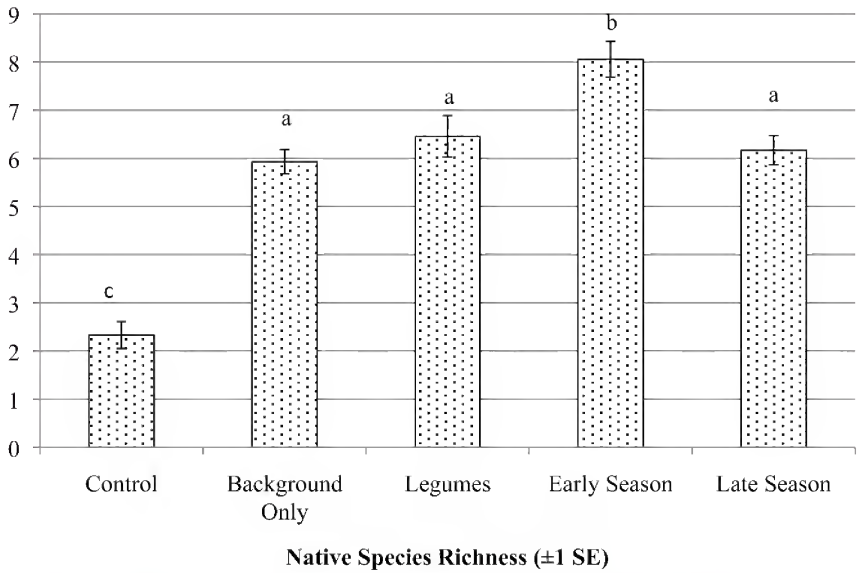

FIGURE 11. Forb Treatment Comparison: Mean species richness encountered in the Native grouping for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

Species richness of the forb grouping averaged 9.14 species among forb treatment groups. The control plots had significantly lower forb species richness (mean = 6.92 species, $p<0.050$) than all forb treatment groups (mean range: 9.08–10.71 species). The early season flowering forbs had significantly higher (mean = 10.71 species, $p<0.050$) forb species richness than all other treatment groups except for legumes (mean = 9.77 species).

Floristic Quality and Diversity

There were significant differences among treatment groups in FQI values ($F_{4,227}=37.96$, $p<0.001$). The mean FQI of forb treatment groups was 7.43. The FQI of the control plots in 2011 was significantly lower (mean = 1.37, $p<0.050$) than that of all forb treatments (mean range: 8.39–9.99). The early season flowering forb treatments had a significantly higher FQI (mean = 9.99, $p<0.050$) than all other treatments except the legumes (mean = 8.89) (Table 12; Figure 12).

There were also significant differences among treatment groups in the Shannon diversity index (H′) ($F_{4,227}=4.86$, $p=0.001$) (Table 12). The mean H′ of forb treatment groups was 1.44. The mean H′ of the control plots and legume treatments (mean = 1.26, 1.33, respectively) was significantly lower ($p<0.050$) than for the early season flowering forb treatments (mean = 1.62) (Table 12; Figure 13).

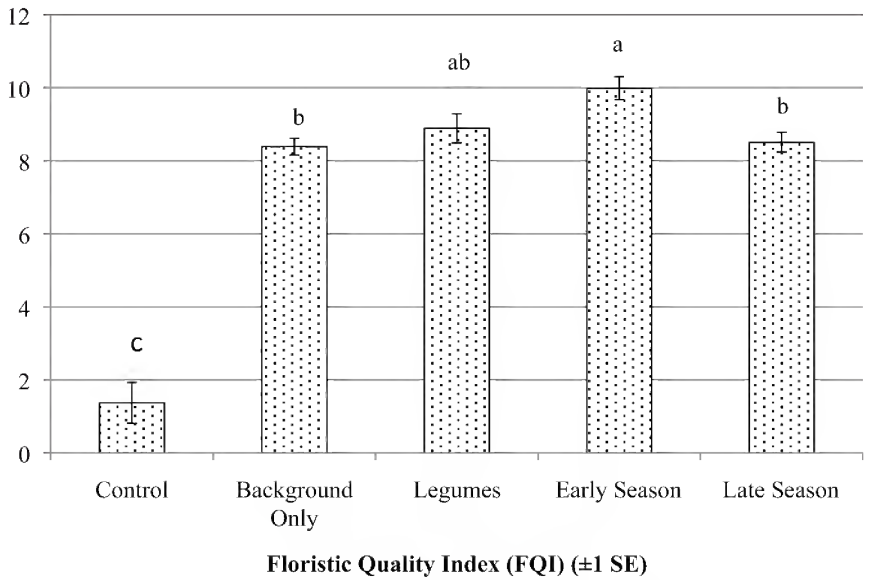

FIGURE 12. Forb Treatment Comparison: Mean FQI for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

TABLE 12. Comparison of the mean floristic quality index (FQI) and Shannon diversity index (H′) values of the forb treatment groups in 2011. Each different measure was tested independently. Values within a column that do not share a superscript letter are significantly different ($p \leq 0.05$) as determined by Tukey post-hoc analysis.

Treatment Group	FQI	H′
Control Plots	1.37(±0.56)	1.26^{a}(±0.09)
Background Forbs Only	8.39^{b}(±0.23)	1.50^{ab}(±0.04)
Legumes	8.89^{ab}(±0.40)	1.33^{a}(±0.06)
Early Season	9.99^{a}(±0.31)	1.62^{b}(±0.05)
Late Season	8.51^{b}(±0.27)	1.50^{ab}(±0.05)

DISCUSSION

Comparison Among Grass Treatment Groups

The percentage cover of resident species was significantly lower in all grass seeded treatments (i.e., no grass, low grass, and high grass), which was likely due to a change in competitive balance (Connell 1983, Fargione and Tilman 2006). The percentage cover of the planted grouping, the species richness of the native and planted groupings, and the FQI were all significantly higher in all the

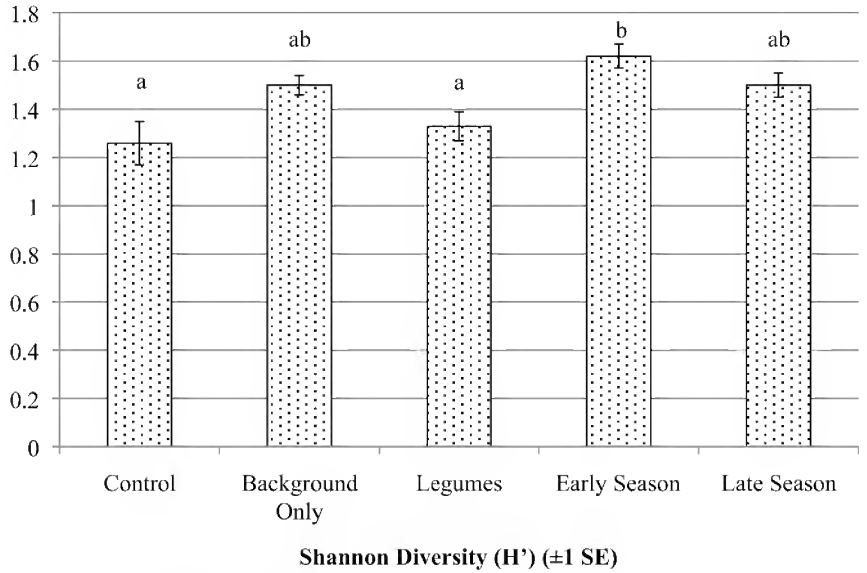

FIGURE 13. Forb Treatment Comparison: Mean H' for several treatment groups in 2011. Bars that do not share the same letter are significantly different ($p \leq .05$) as determined by Tukey post-hoc analysis. Error bars represent one standard error about the mean.

seeded grass plots than in the control plotsm which is due, in part, to an increase in forb species richness in the grass treatments. The baseline data (Table 3) indicates that the community was heavily forb-dominated in 2009. However, native forb species included in the grass treatments further increased the species richness of forbs. Although some grass treatments included forbs, the grass-only treatments also had significantly greater forb species richness than the control plots. Forb species may be benefiting from seeded treatments, because as seeded species grow, they may facilitate the growth of other seeded species around them (Callaway and Walker 1997, Peltzer and Kochy 2001). This may also apply to non-seeded species in the surrounding area. Species currently on or near the site, as well as those species whose seeds are currently dormant in the seed bank, may realize more favorable conditions for germination from the establishment of seeded species, and thereby increase the species richness of forbs.

Comparison Among Forb Treatment Groups

The percentage cover of resident and forb species was significantly higher in the control plots than in the seeded forb treatments. Conversely, the percentage cover of the planted grouping, the species richness of the total, native, planted, and forb groupings, and the FQI were all significantly higher in all seeded forb treatments than in the control plots. Resident cover may be higher in control plots due to reduced competition for available open niche space resulting from

the lack of seed additions in these plots. In contrast to the grass-only treatments, forb cover was higher in treatments that included the addition of forbs. Similarly, planted cover was higher in seeded plots than in the control plots due to the addition of native seed. The species richness of the total, native, and planted groupings and the FQI all increased with the addition of seeded forbs.

Community Responses to Grass Seeding and Forb Seeding Efforts

Treatments with either high or low concentrations of grasses were not significantly different from each other in any measure of species richness other than grass species richness. High grass treatments had significantly higher grass species richness than both low and no grass treatments. Both low and high grass treatments had significantly more total and planted species richness than treatments with no grass.

High grass treatments were more effective in covering more ground and hence had lower bare-ground cover than all other treatments except the highly variable control plots. High grass treatments also had a significantly higher FQI. Diversity (H′) did not differ significantly between high and low grass treatments. However, these treatments were significantly more diverse than all other treatments except for the background only species with no grass treatment.

Although the cover was dominated by grasses, the high grass plots displayed significantly higher native and planted cover than any other treatment. This may be attributed to the inclusion of native species within these planting groups. However, high grass only plots consisting of no planted forb species still exhibited significantly higher native and planted cover than treatments that included forbs and low grass concentrations. High grass treatments also had significantly lower non-native and resident cover than all other treatments.

Native and planted covers were significantly greater in the early season forb and late season forb treatments than in all other forb treatments. Conversely, non-native and resident covers were significantly less in early season and late season forb treatments than in all other treatments.

Early season forb treatments had significantly higher native and planted species richness than all other treatments. Although early season forb treatments had a significantly higher FQI than most other treatments, the FQI of legume treatments were not significantly different. Early season forb treatments also were significantly more diverse (H') than most other treatments except the background only and late season forb treatments.

Facilitation

These results suggest that seed additions may facilitate plant community development in the relatively hot and dry conditions of a sand prairie during the first few years of community development. The increased growth of native species within the grass treatments—especially within high grass seed treatments—as compared with treatments that did not contain grass suggests that seeded grasses may provide suitable microclimatic conditions for native seed

growth, such as shade for reduced heat stress and moisture capture (Plumb-Mentjes 1990; Smith and Huston 1990; Peltzer and Kochy 2001). Similarly, the growth of early season forbs may create greater cover and biomass early in the year which could result in greater shade and moisture capture for late season forb and grass species (Henderson et al. 1988; Brooker et al. 2007; Callaway and Walker 1997).

Similar results indicating a facilitative effect from seed additions have been shown in an experiment where bunchgrasses protected rare plant species during dry years (Greenlee and Callaway 1996) and in other studies where adult plants provided a facilitative response during restoration in arid ecosystems (Maestre et al. 2001; Barchuk et al. 2005). Studies have also attempted to model light and moisture capture of species in dry, harsh environments such as sand prairies. Holmgren et al. (1997) and Smith and Huston (1990) studied the role facilitation plays in community development by modeling the positive and negative effects that "nurse plant" or mature plant canopy cover plays on the establishment of new plants in dry communities. This model shows that the increased availability of water due to facilitation outweighs the detrimental effect of shade (decreased light and photosynthesis) in harsh conditions. Therefore, facilitative effects may outweigh competitive effects of neighboring plants in such environments.

Competition

These results also indicate that competition resulting from seed additions may play a key role in promoting initial community development. Individual resident species respond in different ways to species additions based on their ability to compete for limited resources (Grime 1974, Goldberg and Barton 1992, Smith et al. 1999). However, the overall decrease in resident species growth within the grass treatments—especially within high grass seed treatments—suggests that resident species are at a competitive disadvantage compared to seeded species. Conversely, the overall increased growth of seeded species in the grass treatments suggests that species that are native to the sand prairie may be better able to compete for the limited resources than other early successional, weedy, or non-native resident species such as *Centaurea stoebe* L.

Similar results indicating a competitive effect from grass seed additions have been shown by Jordan et al. (2008), where they found that native grasses were less affected by invasive species. That study found that grasses are relatively insensitive to altered soil biota from invasive plants, and in turn, may shift the competitive balance of restoration efforts on an ecosystem. Similar studies suggest that grasses in prairie restoration enhance overall diversity and reduce exotic species cover (Middleton et al. 2009; Carter and Blair 2012). These studies also indicate that native grasses in restoration may represent an effective management strategy to reduce exotic plant density. Although grass seeding in restorations has been shown to be beneficial, long-term studies have shown that grasses can dominate and exclude other native species over time (Schramm 1990).

These results also indicate that forb additions—particularly a benefit of early season forb treatments and a lack of response from legume treatments—may play a key role in influencing community competitive balance (Howe

1994; Dukes 2001). Additions of early season forb species may provide an initial, early-in-the-year increase in species cover and richness that may in turn directly affect the competitive balance of a community by providing greater resistance or buffer capability to further invasions by invasive species, which in turn may show increased diversity from other native seeded species later in the growing season (Naeem and Wright 2003; Kennedy et al. 2002; Dukes 2001). Contrary to early season forb treatments, legume treatments, although expected to be highly competitive in a nutrient-poor sand prairie ecosystem due to their ability to fix nitrogen, had little effect on plant community composition. Three competitive reasons may explain why we did not see significant establishment of seeded legume species. The dominant and widespread resident legume species *Trifolium arvense* L. may have adverse impacts on seeded legumes. It may suppress seed additions either via interspecific competition, being temporally competitive due to its annual life cycle and producing large amounts of seeds both in the spring and summer, or possibly because nitrogen fixing niche space is already being consumed by *Trifolium arvense* L. (Dukes 2001; Fargione et al. 2003).

Similar experimental results have suggested that seeding of early season forbs within a restoration may provide positive competitive effects to a community. Martin et al. (2005) suggested that restoration seeding efforts that contain early season forbs may be more diverse because these species are better able to coexist with other, later growing seeded species because they come to occupy an early season niche. Thus, early season forbs will ultimately increase diversity, species richness, and other important community variables. Similarly, a study by Foster and Tilman (2003) suggests that early season forb seeding (as part of a complete restoration seed mix) presents an opportunity for transient coexistence through competition-colonization trade-offs, thus allowing many species to coexist at the community scale. Additionally, Seabloom et al. (2003) proposed that a single seeding of native forbs (including early season forb species), even in the presence of high densities of exotic species, may be sufficient to create viable populations of native species in areas that are currently dominated by exotic species.

Similar results showing a lack of legume establishment after seed additions have been found. In a twenty-five year study of prairie establishment following restoration, Schramm (1990) found that legumes did not become a major component of restorations until later phases of restoration succession. However, after legumes became more established in later years of restoration, it was found that they had greater staying power. This could be due to a lack of competitive ability during their early establishment periods (Schramm 1990). This may also be due to interspecific competition for limiting resources among legumes with neighboring plants belonging to the same functional group (Nemec et al. 2013). Most studies that have shown minimal impacts of legumes (species cover and richness) on restorations have been in response to herbivory. Restoration attempts that were not excluded by fencing and had populations of deer or voles saw high mortality or reduced fruiting in legume species due to the grazing preference of these animals for these species (Diaz et al. 2003; Howe et al. 2002).

CONCLUSION

In general, a comparison between seeded treatments and non-seeded control treatments indicates that our efforts have been more successful in the restoration of native sand prairie than would have resulted from succession alone. Restoration attempts displayed a significant decrease in invasive, resident species richness and increased diversity compared to succession. Treatments that included a high concentration of grass and/or an early season forb component had the greatest overall positive impact on plant community development. These treatments exhibited significantly greater diversity and higher floristic quality than most other treatments, and they also displayed less non-native or invasive cover than most other treatments. High concentrations of grasses and early season forbs may therefore play a critical role in initial species establishment of a sand prairie restoration due to the facilitative and competitive advantages they may provide in these harsh environments. However, it remains to be seen if these initially successful communities will have continued success over longer periods of time.

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NOTEWORTHY COLLECTION

NELUMBO LUTEA WILLD. (NELUMBONACEAE): OCCURRENCE OF A RARE PLANT IN WESTERN MICHIGAN CONFIRMED

Previous Knowledge. *Nelumbo lutea* (American Lotus, Lotus-lily, Water Chinquapin) is an aquatic plant of quiet, shallow waters, widespread in the eastern half of North America from Florida, Louisiana, and Texas northward to southern Maine and Massachusetts west to southern Ontario, southern Michigan, Wisconsin, southeastern Minnesota, Iowa, and eastern Nebraska (Crow & Hellquist 2000; USDA, NRCS 2014). *Nelumbo lutea* was documented on the distribution map in Voss and Reznicek (2012) only for Wayne and Monroe Counties, which are adjacent to the well-established and extensive populations in Ohio along the southern shore of Lake Erie from Erie, Sandusky, Ottawa, and Lucas Counties (personal observation), as well as a somewhat disjunct site in southwestern Michigan in Kalamazoo County. According to information from the Michigan Natural Features Inventory (Slaughter, pers. comm.), numerous records from Monroe County are mostly from the Lake Erie marshes. Wayne County records document a population of *N. lutea* adjacent to Lake Erie, as well as from historic sites on the Huron and Rouge Rivers.

The single historical report for Kalamazoo County of a population from Sunset Lake in the town of Vicksburg (Hanes and Hanes 1947) has had its native status questioned (Voss 1985). It had reputedly occurred in the marsh at Sunset Lake in Vicksburg since at least the middle of the 19th century, according to Voss (1985), based on Beal's (1878), citation of a report from a Frank Tuthill of Kalamazoo that the American Lotus must have been "introduced after the country was settled," as the pond in which the plant was found was a mill pond. Beal also mentions a Mrs. Adams, who had lived nearby, who thought *Nelumbo lutea* had grown in the natural pond prior to the establishment of the mill in 1829 (Beal 1878). Beal later stated under his account of this species for all of Michigan, "Perhaps introduced by the Indians" (Beal & Wheeler 1892, Beal 1904). In his article updating the "Flora and Vegetation of Kalamazoo County, Michigan," McKenna (2005, p. 226) stated: "To my knowledge, [the Vicksburg population was] last collected by the Haneses on 11 July 1934." He also reported that a "short, typed report on the flora of the county written by C. R. Hanes (date of preparation unknown, but certainly post 1934) reads 'The Lotus, which formerly grew at Vicksburg . . . .'" McKenna (2005) regarded it as extirpated and probably not native.

Voss (1985) indicated that various attempts had been made in the 1800s to introduce new populations in the Detroit River, with none persisting. I once tried to establish a population of *Nelumbo lutea* in the old reservoir on the campus of the University of New Hampshire, Durham, New Hampshire, with plants taken from a lake in Wisconsin. After I transplanted three tubers, the plants flowered

nicely the following summer, thereafter producing only a few floating leaves for a few years, and then disappearing from the transplanted site. So, if the Vicksburg population was sustained for nearly a century, does it not stand to reason that the occurrence might have been natural? Furthermore, Michigan Flora Online (2011) notes that while *Nelumbo lutea* may have been regarded as perhaps introduced ". . . by Native Americans as a food-plant, . . . there is no documentation one way or the other. Some of our colonies, at least, are assumed to be natural, at the northern edge of the range for the species."

Discussion. The Michigan Natural Features Inventory (2007) lists *Nelumbo lutea* as a rare plant with Threatened state status and S2 (Imperiled) state rank, although it is considered globally Apparently Secure (G4). This documentation of two populations in Ottawa County by herbarium specimens reconfirms the presence of this rare plant in western Michigan, which was first collected by K. Karsten on August 25, 1942; the occurrence was unknown to Voss (1985) in his extensive work on his *Michigan Flora* and was still unknown to Voss and Reznicek (2012) at the time of publication of *Field Manual of Michigan Flora*.

The western Michigan population came to my attention as I was working in the Calvin College Herbarium (CALVIN) in January 2014 on a project to image all of the specimens in the collection. I ran across two sheets of a collection that was made from a population with the label data: "Grand River at Stearns Bayou, Ottawa Co., Coll. *M. Karsten 7a & 7B*, 7-25-*'42*" (Figure 1). Since there was no dot recorded for Ottawa County on the distribution map for this species in Voss and Reznicek (2012), I sent images of Professor Karsten's collection to A. A. Reznicek so that the Michigan Flora Online (2011) website could be updated; Ottawa County has now been added to the list of counties and a dot added to the map.

In August 2014 I set out to relocate the 1942 Ottawa County population documented by Professor Karsten. Using Google Earth I was able to locate a bridge crossing Stearns Bayou on Green Street, Robinson Township. A large population was hiding in plain sight adjacent to an old store/boat livery and launching site called Felix's Landing, with a small sign, "Stearns Bayou," on the side of the building. This large population, in full bloom (Figure 2), was adjacent to a large, dense cattail marsh; the opposite side of the waterway was dominated by *Nymphaea odorata*, but no *Nelumbo lutea* was observed on that side near the bridge. A second visit to the area in August 2015, by canoe, revealed that this population was extremely large, extending the whole length of the channel (both sides) connecting Stearns Bayou to the main flow of the Grand River. Specimens collected now provide present-day documentation of that population, which had been visited by Professor Karsten 72 years earlier, at Stearns Bayou.

Approximately one mile westward, I found a second population along the west shore of Millhouse Bayou, in Grand Haven Township. Here the population was small, consisting mostly of scattered plants at the Township picnic area waterfront and at an adjacent neighbor's dock, growing in water ca. 1 m deep. Specimens were collected to document this additional locality.

In searching for additional information on *Nelumbo lutea* in Michigan, I was surprised to learn that the Michigan Natural Features Inventory (2007) (MNFI) also had internal reports on two other populations in the Grand River system in Ottawa County (Slaughter, pers. comm.): (1) a large population in Grand Haven

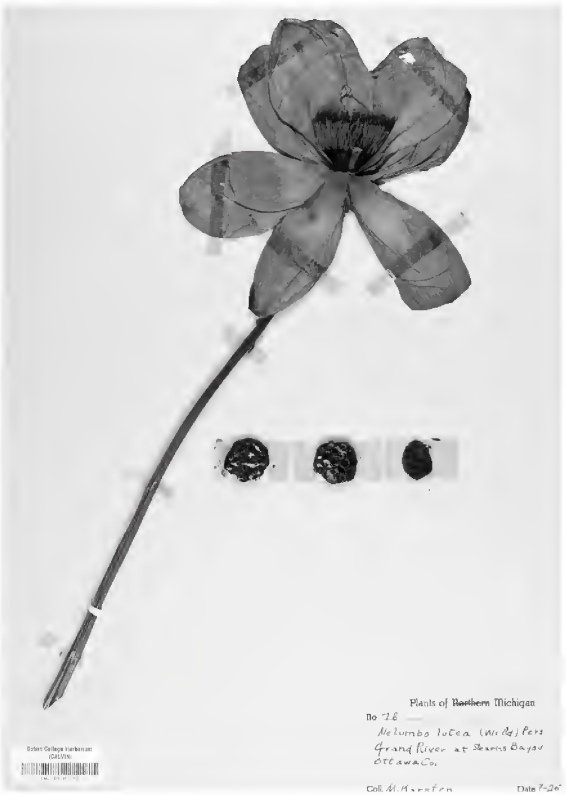

FIGURE 1. Herbarium specimen of *Nelumbo lutea* collected by K. Karsten in 1942 from Stearns Bayou, Ottawa County, Michigan (CALVIN).

Township at the east end of Martinique Island, approximately 3 miles downstream from my Grand Haven Township site; and (2) a large population at the mouth of the Stearns Bayou, about 0.5 mile from my Green Street Bridge site.

Furthermore, the MNFI website has posted a single site in each of Barry, Berrien, and Genesee Counties—all unmapped on Michigan Flora Online (2011). There is clearly a need for documentation of populations of rare plants by depositing voucher specimens in our major state herbaria, thereby assuring that there will be a permanent record. Information from internal records at MNFI (Slaughter, pers. comm.) for Genesee County indicates a healthy population on Fenton Lake in 2005; in southwest Michigan, the Barry County population at Algonquin Lake was surveyed in 2004; and a report for Boyle Lake, Berrien County, first reported in 1934, indicates that *N. lutea* was "Apparently observed in 1981 . . . with no other data provided."

Diagnostic Characters: *Nelumbo lutea* is readily recognized, being a large aquatic plant producing large circular leaves with the petiole attached in the center of the blade (peltate), the leaves standing erect and emergent, and as floating leaves (especially in deeper water or earlier in the growing season). Populations are conspicuous in late summer (Figures 2 and 3), as the flowers are pale-yellow

FIGURE 2. View of present-day population of *Nelumbo lutea* at Green Street Bridge, Stearns Bayou, Robinson Township, Ottawa Co., Michigan. Photo by Garrett Crow.

FIGURE 3. *Nelumbo lutea* population at mouth of Stearns Bayou. Photo by Garrett Crow.

to sulfur yellow, with flowering often peaking in mid-August; the petals and stamens are numerous, and the numerous carpels are separate and imbedded in a broad, flat-toped receptacle (Figure 4). Large populations can readily be spotted from a distance. The plants readily spread by rhizomes, often forming large populations that are near monocultures. Large tubers are produced in late summer, and were eaten by Native Americans. The fruits are round nutlets with a hard

FIGURE 4. Flower of *Nelumbo lutea*. Photo by Garrett Crow.

FIGURE 5. Maturing fruiting receptacle (torus) of *Nelumbo lutea* with developing nutlets in cavities. Photo by Garrett Crow.

outer wall, remaining in or shed from cavities in the mature woody receptacle (Figure 5).

Nelumbo lutea is the only species of the Nelumbonaceae native to North America and is closely related to the well known pink- or white-flowered *N. nucifera* (Sacred Lotus or Oriental Lotus) native to eastern Asia and northern Australia and commonly cultivated in water gardens. It occurs very sporadically as naturalized populations in the southeastern United States (Wiersema 1997). *Nelumbo lutea* and *N. nucifera*, the only two taxa in the Nelumbonaceae, are very closely related, and it has even been proposed that the two be regarded as a single species with two widely disjunct subspecies; the name for our taxon would then be *N. nucifera* subsp. *lutea* (Willd.) Borsch & Barthlott. But I am retaining the two species viewpoint, following the nomenclature of the taxonomic treatment of *Flora of North America North of Mexico* (Wiersema 1997), *Field Manual of Michigan Flora* (Voss and Reznicek 2012), and *Aquatic and Wetland Plants of Northeastern North America* (Crow and Hellquist 2002).

Specimen Citations. Michigan, Ottawa Co., Grand River at Stearns Bayou. July 25, 1942. *M. Karsten 7a & 7B* [CALVIN002319 & CALVIN002321]; Robinson Township, Stearns Bayou at Green Street Bridge, Felix's Landing; backwater off Grand River; large population adjacent to boat launch on northeast side of bayou. August 6, 2014, *Garrett E. Crow 10831*, and 17 August 2015, *Garrett E. Crow 10978* (CALVIN, MICH, MSC); Grand Haven Township, Millhouse Bayou at township picnic area on Bignell Road; backwater off of Grand River; small population with scattered plants in shallow areas near shore. August 6, 2014. *Garrett E. Crow 10836* (CALVIN, MICH, MSC).

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Special thanks are extended to Brad Slaughter for providing information from the Michigan Natural Features Inventory on *Nelumbo lutea*, as well as for providing suggestions on improving the manuscript. Anonymous reviewers also provided constructive comments. Thanks are extended to David and Charylene Powers for providing the opportunity to examine the population by canoe, as well as to my field assistant, Charlyn Crow.

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——Garrett E. Crow
Professor Emeritus, University of New Hampshire
Visiting Scholar, Calvin College
Visiting Research Botanist, Michigan State University Herbarium
garrett.crow@unh.edu

NOTEWORTHY COLLECTION

THE FIRST RECORD FOR MICHIGAN OF *GALIUM PEDEMONTANUM* (RUBIACEAE)

Significance of the Report. The first recorded occurrence in Michigan of this widespread European annual.

Previous Knowledge. The earliest reported collection of *Galium pedemontanum* (Bellardi) All. (Foothills bedstraw) in the United States is from Kentucky in 1933 (Sanders 1976). By 1976, it had already spread to Arkansas, Illinois, Missouri, Ohio, Oklahoma, Tennessee, Virginia, and West Virginia (Sanders 1976). Currently, its distribution includes the additional states of New York, Pennsylvania, North Carolina, South Carolina. Georgia, Mississippi, Louisiana, Texas, Indiana, Nebraska, and the Pacific Northwest states of Washington, Oregon, Idaho, and Montana (USDA, NRCS 2015). The nearest location to the one reported here for Michigan is about 23 km south-southeast in St. Joseph County, Indiana (Swink and Wilhelm 1994).

Discussion. *Galium pedemontanum*, a slender, sprawling, annual plant with retrorsely-barbed stems, was probably an accidental introduction into the flora of the United States. Using collection data from herbarium specimens, Sanders (1976) traced the spread of the species from its first reported locality, showing that it had been transported in all directions from Lexington, Kentucky. Going a step further, Sanders also predicted the future distribution of this species by identifying and using the soil and climate characteristics of its European habitats to delineate similar sites in the USA. In Europe, *G. pedemontanum* is found where the mean annual temperature is 8–18°C, the mean annual rainfall is 40–150 cm., and the soils, which have developed under forests, are acidic (Sanders 1976). With a mean annual temperature of 7–9°C, a mean annual precipitation of 76–81 cm, and soils mapped as fine sands which are acidic to neutral (USDA, NRCS 2013), the climate and soils of southwestern Michigan are similar to Sanders' European climate and soils. Interestingly, his map shows southwestern Michigan at one of the northern edges of the potential USA distribution of *G. pedemontanum*.

I spotted the population of *Galium pedemontanum*, the dozen plants of which were scattered over an area of approximately 2 m^2, because the elongated, lax, pale stems stood out against the dark green of the lawn grasses in the middle of summer 2015. In early May 2016, numerous plants, now verdant and semi-erect, were growing in a 25 m^2 area at the same location, but now associated with other spring-flowering introduced species (e. g., *Scleranthus annuus* L., *Veronica arvensis* L., *Arenaria serpyllifolia* L., *Cardamine hirsuta* L., *Lamium purpureum* L., *Oxalis stricta* Jacq., and *Myosotis stricta* Roem. & Schult.). Because this location is close to a road, I suspect that the seeds of *G. pedemontanum* may have been carried in by vehicle traffic.

Recent studies (e.g., Ehrendorfer and Barfuss 2014) suggest that *Galium*

FIGURE 1. *Galium pedemontanum* at the collection site.

pedemontanum, along with several other Eurasian species, should be segregated as the genus *Cruciata*. In that case, our species would be known as *Cruciata pedemontana* (Bellardi) Ehrend.

Diagnostic Characters. *Galium pedemontanum* is one of only two annual species among the 21 species of *Galium* now known from Michigan (Voss and

Reznicek 2012). It is a weak-stemmed, scrambling plant with noticeably scabrous stems. It may be distinguished from the other annual species in Michigan, *Galium aparine* L., by its greenish-yellow flowers with glabrous ovaries and its four small ovate leaves per node, and from *Galium verum* L., the other yellow-flowered Michigan bedstraw, by its shorter leaves. Below are the key characters that can be used to separate *Galium pedemontanum* from *Galium aparine* and from *Galium verum*.

Galium aparine—leaves 6–8 at each node, 1–8 cm long, flowers greenish white, fruits hispid.
Galium pedemontanum—leaves 4 at each node, 3–11 mm long, flowers dull yellow, fruits glabrous
Galium verum—perennial, leaves 4 at each node, 1–5.4 cm long, flowers yellow, fruits glabrous.

Specimen Citation. Michigan, Berrien County, Sawyer (N41° 53.681′; W86° 36.585′). In sandy soil of open lawn with *Poa pratensis, Medicago lupulina* and *Oxalis stricta*. July 27, 2015, *Tatina s.n.* (MICH).

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——Robert Tatina
Dakota Wesleyan University
Mitchell, South Dakota 57301
rotatina@dwu.edu

BOOK REVIEW

Emmet J. Judziewicz, Robert W. Freckman, Lynn G. Clark, and Merel R. Black. 2014. *Field Guide to Wisconsin Grasses*. The University of Wisconsin Press, Madison. ix + 396 pp. ISBN 978-0-299-30134-7. Paperback. $29.95 (also available as an e-book for $24.95).

Andrew L. Hipp. 2008. *Field Guide to Wisconsin Sedges: An Introduction to the Genus* Carex *(Cyperaceae)*. The University of Wisconsin press, Madison. xi + 265 pp. ISBN 978-0-299-22594-1. Paperback. $29.95 (also available as an e-book for $19.95) (out of print in hardcover).

A significant component of nearly all ecosystems consists of the graminoids—grasses (family Poaceae), sedges (family Cyperaceae), and rushes (family Juncaceae)—and in ecosystems such as prairies and wetlands, where they are particularly numerous in species, they also form the bulk of the biomass. Accurate identification of graminoids is therefore essential for any floristic study, ecological survey, restoration project, or similar endeavor. Unfortunately, these ecologically and floristically important plant groups are rarely included at all in wildflower guides and other popular references. In many places, one must depend on standard floras that, although including graminoids, often do little to make identification easy, relying primarily on keys, only sometimes containing descriptions (often short), and often lacking any illustrative materials. Such floras generally give short shrift to explanations of the unique structural features of these plants and the specialized terminology necessary for understanding and identifying them. For an expert, this may be no great barrier, but for the novice, it creates a steep learning curve indeed.

Wisconsin is fortunate in having two up-to-date guides that cover all species of two major groups of graminoids. The first book under review covers all of the grass species in Wisconsin, and its companion covers all of the species of *Carex*, which, with 157 species, is the largest genus of sedges in Wisconsin. Both of these volumes go to great lengths to make life easy for the novice and expert alike, while still being complete and technically accurate.

Let's look at the *Field Guide to Wisconsin Grasses* first. The book starts with a detailed introduction to grass structure, replete with photographs of such things as culms (stems), leaves (including sheaths, auricles, and ligules), all the details and variations of inflorescences—spikelets and their various arrangements, glumes, and lemmas—and finally the tiny flowers. There is a key to genera, accompanied by detailed line drawing of the relevant characters in the margin. The key is preceded by several very useful features: (i) a discussion of what to look for in identifying an unknown grass, with reference to several critical characters, (ii) lists of grasses with unusual characters, and (iii) a list and short descriptions of the ten commonest grass species in the state.

The main part of the book is devoted to a species-by-species account,

arranged alphabetically by genus. Each genus has a short description and, if there are more than one species in Wisconsin, a key to species. The individual species treatments vary somewhat, depending on what is felt to be needed in each case. There may be a short description; more commonly, however, a brief statement of characters contrasting the species with related, or similar, species, is given. Habitat and distribution data is generally included, as well as a county distribution map. Most usefully, most accounts contain both photographs and line drawing of important characteristics.

Field Guide to Wisconsin Sedges takes a somewhat different approach. Although there is a brief discussion of structure and terminology, the principal early portion of the book is a breakdown of the genus Carex into two subgenera and then into artificial groupings within each subgenus. Several keys are provided, ultimately reaching all species of Carex in Wisconsin. These are followed by a lengthy treatment of the several sections of the genus, each with a description, and a discussion of the species within the section. All of this preliminary material is based on the sound idea that learning sedges is made much easier by grasping the subgenera and sections before trying to tackle identification of individual species. The individual species treatments, which constitute about half of the book, treat 55 common species in detail. Each treatment contains a brief diagnostic statement, a detailed description of the plant, a description of the habitat and range within the state, and, significantly, a detailed discussion of similar species and discussions between them. Each treatment is also accompanied by an illustration in ink and watercolor by the artist Rachel D. Davis containing both a view of the habit of the plant and a detailed view of the inflorescence or individual perigynia (the closed sac-like structures that enclose the female flowers). Distribution maps of all species of *Carex* in the state, prepared by Merel R. Black and Theodore S. Cochrane, are collected in an appendix.

A particularly valuable feature of both volumes under review is a discussion of the plant communities in which species of grasses or sedges occur, along with lists of species for each habitat. For the novice in particular, this is a very useful aid in narrowing down the process of identifying an unknown. This portion of the sedge guide was prepared by Theodore S. Cochrane.

Because a large percentage of the species treated in both of the field guides tend to occur in suitable habitats throughout the western Great Lakes region, these volumes will be valuable resources for students and investigators alike throughout this region.

——Michael Huft

ANNOUNCEMENTS

ISOBEL DICKINSON MEMORIAL AWARD RECIPIENTS

Congratulations to **Emily Huizenga** and **Christopher Bouma**, who were the recipients of the Isobel Dickinson Memorial Award for **best student-authored paper** published in *The Michigan Botanist*, Volume 52. The selected paper was entitled "Assessing a Reconciliation Ecology Approach to Suburban Landscaping: Biodiversity on a College Campus" by Christopher Bouma, Emily Huizenga, and David Warners, The Michigan Botanist 52: 93–104. We acknowledge the Michigan Botanical Club–Dickinson Award Committee for evaluation of the student papers and the Michigan Botanical Foundation for funding this award.

REVIEWERS FOR VOLUMES 53 AND 54

I wish to thank the following people who reviewed manuscripts for Volumes 53 and 54 of *The Michigan Botanist*. Their comments were important, both to the authors and to the editor, and their efforts, which are essential to maintaining the high quality of the journal, are greatly appreciated.

Dennis Albert
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Anton A. Reznicek
Dennis Riege
Paul Rothrock
Michael Rotter
Lisa Schulte
Gordon Tucker
Dennis Woodland
John Wiersema

INSTRUCTIONS TO AUTHORS

Refer to http://quod.lib.umich.edu/m/mbot/submit for more detailed instructions, especially for formatting, style conventions, literature cited, and voucher specimen requirements. Please contact the editor with any questions.

1. Create text in 12-point Times New Roman font and double space paragraphs throughout. Research articles should be organized as follows: Title, Author(s) and address(es), Abstract with up to 5 keywords, Introduction, Materials and Methods, Results, Discussion, Acknowledgements, Literature Cited, Tables, Figure Legends, and Figures. Sections may be omitted if not relevant. All pages should be numbered.

2. For noteworthy collections, manuscripts should be formatted as follows. The title, "Noteworthy Collections," should begin each submitted manuscript, followed on the next line by the State or Province for the species reported. The next line should list the taxon of interest using the following format: Species Author(s) (Family). Common name. The rest of the manuscript should include the following named sections: (i) Significance of the Report, (ii) Previous Knowledge, (iii) Discussion, (iv) Diagnostic Characters (if desired), (v) Specimen Citations, (vi) Acknowledgements (if desired), and (vii) Literature Cited. Each of these sections is largely self-explanatory; however, the "Significance of the Report" section should be limited to a brief sentence or phrase indicating the significance of the collection(s), and this may be expanded upon in the "Discussion" section; the "Specimen Citations" section should include the relevant label data from the voucher specimen(s) including location data, collector(s), collection number, etc., as well as the Index Herbariorum acronym(s) of the herbarium or herbaria where the specimen(s) are deposited. The manuscript should end with the name and address of the author(s).

3. Non-research articles, such as book reviews, letters to the editor, notices, biographies and other general interest articles can be formatted as general text without the specific sections listed above. However, literature cited and any tables or figures should be formatted as described below.

4. Create tables either as an MS Word table or using a tab-delimited format. Each table is to be submitted as a separate file. Table captions should be placed at the top of the table. Any footnotes should appear at the bottom of the table. Please do not insert tables within the body of the text.

5. Send each figure as a separate file in a high-resolution format—eps, jpg, or tif. Figures like bar graphs that gain their meaning with color won't work—use coarse-grained cross-hatching, etc. Create figure legends as a separate text file, and the typesetter will insert them as appropriate. Please do not insert the figure in the body of the text file.

6. Citations: Please verify that all references cited in the text are present in the literature cited section and vice versa. Citations within the text should list the author's last name and publication year (e. g. Smith 1990). For works with more than 2 authors, use "et al.", and separate multiple citations with a semicolon.

7. Literature Cited: List citations alphabetically by author's last name. The first author's name is to be listed with surname first, followed by initials (e.g. Smith, E. B.), and subsequent authors are to be listed with initials first.Separate author's initials with a single space. The year of publication should appear in parentheses immediately before the title of the citation. The entire journal name or book title should be spelled out. Please put a space after the colon when citing volume number and page numbers.

8. Italicize all scientific names. Voucher specimens must be cited in floristic works and in any other study whose results depend on the identity of the plant(s). Papers citing plant records without documenting vouchers are generally not acceptable.

9. Manuscripts must be submitted electronically to the email address of the editor. All manuscripts will be reviewed by at least two referees.

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On the cover: *Nelumbo lutea*, Beulah Beach, Ohio. Shallow pond behind the beach at mouth of creek entering Lake Erie. *Crow 10830* (CALVIN, MICH, MSC). Photo by Garrett E. Crow.